TESIS DOCTORAL Ecología de laderas restauradas de la minería de carbón a cielo abierto: interacciones ecohidrológicas Luis Merino Martín 2010

ECOLOGÍA DE LADERAS RESTAURADAS DE LA MINERÍA DE CARBÓN A CIELO ABIERTO: INTERACCIONES ECOHIDROLÓGICAS

ECOLOGY OF RESTORED SLOPES FROM OPENCAST COAL MINING: ECOHYDROLOGYCAL INTERACTIONS

TESIS DOCTORAL

Luis Merino Martín
2010

Universidad de Alcalá
Departamento de Ecología

Universidad de Alcalá
DEPARTAMENTO DE ECOLOGÍA

Ecología de laderas restauradas de la minería de carbón a cielo abierto: interacciones ecohidrológicas

Memoria presentada por D. **Luis Merino Martín**
para optar al grado de Doctor por la Universidad de Alcalá

Directores
José Manuel Nicolau Ibarra
Tíscar Espigares Pinilla

Alcalá de Henares, septiembre de 2010

Universidad de Alcalá
DEPARTAMENTO DE ECOLOGÍA

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

José Manuel Nicolau Ibarra, Profesor Titular del Departamento de de Agricultura y Economía Agraria de la Universidad de Zaragoza,

HACE CONSTAR:

Que el trabajo descrito en la presente memoria, titulado **“Ecología de laderas restauradas de la minería de carbón a cielo abierto: interacciones ecohidrológicas”**, ha sido realizado bajo su dirección por D. Luis Merino Martín el Departamento de Ecología de la Universidad de Alcalá, y reúne todos los requisitos necesarios para su aprobación como Tesis Doctoral.

Alcalá de Henares, a 16 de septiembre de dos mil diez,

Dr. José Manuel Nicolau Ibarra
Director de la Tesis

Universidad de Alcalá
DEPARTAMENTO DE ECOLOGÍA

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

Tíscar Espigares Pinilla, Profesora Titular del Departamento de Ecología de la Universidad de Alcalá,

HACE CONSTAR:

Que el trabajo descrito en la presente memoria, titulado **“Ecología de laderas restauradas de la minería de carbón a cielo abierto: interacciones ecohidrológicas”**, ha sido realizado bajo su dirección por D. Luis Merino Martín el Departamento de Ecología de la Universidad de Alcalá, y reúne todos los requisitos necesarios para su aprobación como Tesis Doctoral.

Alcalá de Henares, a 16 de septiembre de dos mil diez,

Dra. Tíscar Espigares Pinilla
Directora de la Tesis

Universidad de Alcalá
DEPARTAMENTO DE ECOLOGÍA

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

Julio Camargo Benjumeda, Profesor Titular y Director del Departamento de Ecología de la Universidad de Alcalá,

Hace constar:

Que el trabajo descrito en la presente memoria, titulado **“Ecología de laderas restauradas de la minería de carbón a cielo abierto: interacciones ecohidrológicas”**, ha sido realizado por D. Luis Merino Martín dentro del Programa de Doctorado Cambio Global y Desarrollo Sostenible adscrita al Departamento de Ecología de la Universidad de Alcalá, y reúne todos los requisitos necesarios para su aprobación como Tesis Doctoral.

Alcalá de Henares, 16 de septiembre de dos mil diez.

Dr. Julio Camargo Benjumeda
Director del Departamento

A mis padres,
sin esa raíz, que sería de mí...
y a Nico y Tíscar,
troncos por los que ha sido un gusto crecer.

Caminante, no hay camino

Caminante son tus huellas
El camino nada más;
caminante no hay camino
se hace camino al andar.
Al andar se hace camino
y al volver la vista atrás
se ve la senda que nunca
se ha de volver a pisar.
Caminante, no hay camino
sino estelas sobre el mar.
¿Para qué llamar caminos
A los surcos del azar…?

Antonio Machado

Agradecimientos

Hay treinta radios que forman una rueda,
y que convergen en el eje,
pero es el vacío el que permite a la rueda moverse.
Utilizamos la arcilla para hacer recipientes,
pero es el vacío de su interior,
lo que las hace útiles.
Construimos con maderas una casa con una habitación,
separada con puertas y ventanas,
pero es el vacío de su interior lo que buscamos.
Por lo tanto, de lo que "es" obtenemos ventajas.
pero en lo que "no es" está la utilidad de las mismas.
Tao Te King
Lao Tse

De lo que hay aquí en esta Tesis escrito, he obtenido ventajas, pero de todo lo que no hay escrito, he obtenido la utilidad de las mismas. Gracias a todos. Como siempre, de lo más bonito, no hay señales; nada queda de esas largas conversaciones camino a Teruel, nada de esos temblores bajo el viento duro y nevoso del invierno Turolense, nada queda de esas risas discutiendo sobre lo que esconde El Moral, nada queda, nada más…que lo más importante, el mejor recuerdo que encoje el corazón.

Gracias Nico, por enseñarme a dejar de *mirar*, y ponerme en el camino de *observar*. Por enseñarme a observar detrás de lo que se ve, en la Naturaleza y en las personas. Por acompañarme de todas las formas posibles en esta, nuestra carrera de fondo. Por esos empujones para creer en mí y en mi trabajo y por tantas horas dedicadas a nuestra Tesis. Por esas largas conversaciones camino de Teruel, por esos largos paseos por El Moral, por esos abrazos y esos baños en el río. Muchas, muchas gracias por cruzar ese turbulento río. Nico, gracias por darnos la mayor de las alegrías, de verdad que sin ti no hubiera podido.

Gracias Tíscar, por enseñarme tantísimo, de verdad. Gracias por enseñarme a hacer las cosas con pasión y a creer en lo que hacemos. A darle la importancia que tiene cada cosa y a confiar ante las dificultades. A dar lo que tenemos y lo que somos. A trabajar con precisión, y con corazón. Gracias por tantísimo apoyo y cariño. Y siempre, siempre, con esa sonrisa en la cara y con esas ganas de reír a la vida.

Gracias a los dos, sois un equipo genial, os voy a echar mucho de menos. Una vez, no se me olvidará, me dijeron que el mejor maestro, no enseña sólo lo que sabe, sino lo que es. Y yo siento de corazón, que he tenido los mejores maestros.

Gracias Mariano, cómo no, mi hermano mayor en esto de la ciencia. Cuántas horas hemos pasado en el campo, trabajando durísimo, haciendo (como dice Lucía) de *da Vinci* de la ciencia; haciendo cemento, construyendo zanjas, cortando con la radial, instalando vallas…pasando mucho calor, y mucho frío.

En este equipo en el que aparecí un día, tan excepcional, académicamente y en lo personal, he intentado humildemente seguir vuestra maravillosa estela.

Y luego vino Silvia, gracias por tantísima ayuda en la instalación de la estación experimental y por ese año tan genial que compartimos en Utrillas de tantísimo trabajo de campo. Gracias también a mi compañera C-15, que tantísimos kilómetros y días de campo hemos pasado juntos (gracias a ti, me hice amigo del mecánico de Monreal del Campo). Gracias pueblo de Utrillas, Paco, el alcalde y operarios del Ayuntamiento y mi querida vecina de *Los coloritos*, que siempre estaba ahí para ayudar, gracias a todos por hacer de Teruel un lugar más acogedor.

Gracias *Cerro del Moral*, muchas han sido las horas que he pasado muestreándote, andándote y pensándote. Ha sido un gusto pasar tantas horas con mi compañera "soledad" escuchando cómo se infiltraba el agua de lluvia, el ruido del oleaje de los sembrados y el sonido de las vainas de las genistas al abrirse (qué gozada). Acabar embarrado hasta las cejas en cada lluvia no siempre era agradable, pero ahora lo echo de menos. Gracias Moral por la "*bromita*" que me gastaste dejando de lloverte justo cuando yo más me llovía, aún así, gracias, me vino muy muy bien. Gracias también *Chorrero* por tantas horas de descanso y de frescor en el verano y tantos gélidos y efímeros baños.

Gracias profesores, compañeros y amigos del Departamento de Ecología de la Universidad de Alcalá, que ha sido mi casa durante muchos años, en la que he estado tan a gusto y ha sido un placer vivir mi desarrollo académico. Gracias a la Universidad por darme ese regalo cuando me concedió la beca para realizar la Tesis Doctoral.

Gracias Ana, mi mamá del departamento. Buenas personas son las que hacen el bien, personas magníficas son las que lo hacen porque les sale de dentro. Ana, gracias por ser una magnífica persona, por preocuparte por tus "*hijos del departamento*" por esos consejos que sólo tú puedes dar y que a muchos nos vienen tan bien. Gracias por todo lo que nos enseñas siendo cómo eres, preocupándote por nosotros y por ese ejemplo de vida que nos das. Voy a echar mucho de menos saludarte y hablarte cada día.

Gracias Lorenzo por estar siempre listo para ayudar y escuchar a pesar de las mil tareas que siempre te rodean. Me alegra muchísimo verte tan feliz…Gracias Gonzalo y Javi, por estar ahí siempre dispuestos a ayudar. Gracias Lilo por preocuparte tanto por mí, porque esté bien, porque siempre tienes una sonrisa y estás lista para escuchar. Gracias Vir, por muchas cosas. Por esos viajes, por ese frío amanecer del Cañón del Colorado, por tantas horas de escucha de *truchos*, por tantos consejos y por ser mi "alma gemela" del departamento. Quién nos iba a decir cuando te fuimos a visitar a Salamanca para que nos enseñaras el TDR que luego acabarías tan cerquita, ya siempre cerca. Gracias Óscar y Marta por tantos ratos juntos, por compartir tanto conmigo, por tantos cafés, por escuchar tantos lloros y por darme tantos consejos. Gracias por tantísima ayuda y afecto. Gracias Nacho por esos días que te viniste a las simulaciones de lluvia y esos cafés en el banco hablando de nuestras cosas. Gracias Virginia por escuchar y por hacerme valer en los momentos difíciles. Por darme tanto apoyo y preocuparte por mi bienestar. Por enseñarme a hacer las cosas porque salen de dentro. Gracias Txiqui por esas vueltas a Avenida de América, por ser mi compañera de inseguridades de Virgo y por todos los consejos que derrochabas para animar en los momentos más duros. Gracias Lucía, por ser mi oreja en los momentos más duros, por secar lágrimas y escuchar y aconsejar cómo sólo tú sabes. Por fomentar siempre el buen rollo y por enseñarme a creer en lo que hacemos. Gracias también por todo lo que hay de ti en esta Tesis con las elaboradas correcciones del inglés. Gracias Luis por tu ayuda con la estadística, siempre ahí para responder y ayudar ¡eres una máquina! Gracias Alberto y Nuria por esos bonitos días en el campo. Alberto, por ser mi compañero del *limbo*; ahora, Alberto, sólo quedas tú. Gracias Edu por la ayuda con los multivariantes y por esos truchos en las comidas en el césped. Gracias Marga por enseñarme tantas cosas sobre la vida, ¿cómo está tu corazón? Al final, es lo importante. Gracias Pedrarias por esas historias de Nicaragua Nicaragüita. Gracias a las posteriores incorporaciones del departamento, Cristina, Norberto, Noelia, Belén, porque siempre estáis de buen rollo y con energía. Gracias Sara por la ilusión que siempre muestras al reencontrarnos y esas riquísimas cenas con salchichas de tofu.

Gracias Bárbara, mi Bartoli, mi hermana del departamento, mi compañera de bartoladas. Por enseñarme tantísimo de estadística ¡cuánto sabes! y ser una buenísima correctora. Pero sobre todo gracias por todo lo que me has enseñado sobre las cosas más importantes, sobre la vida, en la que no hay "p" que valga. Dar todo lo que tienes, trabajar para los demás sin esperar ovación, creer y defender los valores de cada uno son conceptos que no se encuentran en ningún paper y que sólo se pueden aprender desde el ejemplo, gracias por enseñarme tantas cosas. Ojalá podamos seguir compartiendo por mucho tiempo nuestras alegrías y nuestras penas. Cada vez que veo un sugus me acuerdo de ti y de todo lo que me has enseñado.

Gracias Kike, por todo, que es mucho y aquí no cabe. Por esa suerte y ese apoyo que es tener un amigo como tú a mi lado. Gracias por venir a Utrillas a compartir El Moral. Por todo lo que hay en esta Tesis de ti, la maquetación es cosa tuya. Pero cómo bien sabes, eso, no es lo más importante que hay aquí escrito. Por nuestros viajes y las largas horas de charlas, risas, lloros y abrazos. Por tu apoyo incansable. Por ser mi compañero de andanzas y peripecias. Gracias Sancho, por decirme siempre que no eran gigantes sino molinos lo que allí había. No sé si alguna vez encontraremos *Nunca Jamás*, pero, ¿y lo que disfrutamos en la búsqueda? Gracias por improvisar conmigo el guión definitivo, seguro que seguiremos escribiendo muchas más historias de película, de la nuestra. ¡Estoy impaciente! Gracias, mai fren, de todo corazón.

Gracias familia, padres, porque siempre estáis ahí, apoyándome cuando me estiro para abrazarme al tallo de otra rama. Gracias por confiar en mí, y por aceptarme en cada decisión que no era la que teníais pensada para mí. Gracias hermanos, por lo que hay aquí dentro de esta Tesis de vosotros (que es mucho) y por lo que no hay, gracias por ayudarme tantísimo, siempre. Gracias Juan y Manuel, por esos abrazos y besos y por esos dibujos que me acompañaban cada vez que echaba de menos el no poder estar con vosotros durante los veranos.

Gracias "*Estrella Polar*", por tantos caminos recorridos, y los que nos quedan. En especial a vosotros mis "chavales" de la Tropa Makalu, porque con vosotros descubrí mi vocación de aprender enseñando "en" y "sobre" la Naturaleza. Gracias Malak, Judith, Rasty, Juanjo, Fer, Trina, Julito, Kibú y Ka por ser mis compañeros de "promesas" y de sueños. Por haberme enseñado tantísimo sobre la vida y por haberme acompañado siempre. Gracias Kibú, por tu infinito apoyo y por tu compresión.

Gracias amigos de "*Mundo perfecto?!*" por escucharme tanto y compartir tantísimo conmigo. Por esos días de campo, de fiestas y de charlas. Y por comprenderme tantísimo durante la elaboración de esta Tesis. Gracias Carol, Jaime, Ana, María, Sara, Patri, Henar, Javito, Berta y Raúl. Gracias Isabella por la parte del caminito que compartiste conmigo.

Gracias amigos de *las Zorreras* por esos veranos de bicis y toallas húmedas, por esas tardes en la "roca" mirando pasar el tiempo, por ese "helicóptero" encina, y ese "barco" montón de ladrillos, por esas tardes esperando que se pasara la hora de la digestión y por esas lágrimas de San Lorenzo. Con vosotros caí en cuenta de lo que me gustaba la Naturaleza. Gracias a Carlitos por esas simulaciones de lluvia que sin ti hubieran sido imposibles, por ese vaso de precipitados que rompiste con el culo y por esas horas rodando sobre la bici.

Gracias amigos del *Pilar* por cada discusión sobre lo distintos que somos y lo distinto que vemos la vida. Por quererme siendo ese *"hippy"* que disfruta del campo, de bañarse en un río y de andar con un macuto. Gracias David y Edu por comprenderme tantísimo.

Gracias Camilo y Paula, por toda la ayuda y el cariño en la *arid* Arizona. Sentíamos que allí, en mitad del desierto, nacía algo que duraría mucho, y el regalo de nuestra siguiente etapa en La Gloria nos lo ha confirmado. Colombia y Guatemala serán mis próximos destinos y quién sabe por cuánto. Gracias Dave, Jason, Darin y el resto de gente del Terrestrial Ecology Lab por todo lo que me enseñaron sobre la ciencia (que fue muchísimo) y por regalarme esa increíble experiencia al otro lado del mundo. Y gracias Anna por ser mi compañera y amiga *visiting scholar* en el desierto y darle ese toque mediterráneo.

Gracias compañer@s del Colegio San Ignacio de Loyola durante mi preciosa etapa en él. Muchísimas gracias Ana, por cogerme de las orejas y plantarme frente a la pared de la vocación. Por esas risas de todas las mañanas antes de que empezáramos las clases. Y cómo no, gracias a nuestros Abdelilah y Abdu, a Houston y sus tablas de multiplicar y a nuestros pequeños serbios de 2 metros, Stefan e Iván, por ese "Quiero aprender" y por demostrarnos que los niños tienen mucha razón, mucha fuerza y mucho que enseñarnos. (Tengooooo…y lo que tengo lo mantengo a base de amor y fe).

Gracias Anabella, Diego Martín y Joan Manuel por haberme regalado una etapa final de Tesis tan preciosa. Gracias Anabella, por esa mirada al escuchar y por esas buenísimas cenas (¡hay que ver cómo te quedan los garbanzos!). Gracias Joan Manuel y Diego Martín por enseñarme tanto sobre cómo hacer las cosas con pasión y con ilusión.

Y cómo no, gracias a ése ángel de la guarda, que me ha ayudado tanto durante esta Tesis, dándome la mayor de las suertes: todas estas personas que aquí aparecen y a las que estoy tan agradecido...Gracias de todo corazón.

Volveré a encontrarme con vosotros,
volveré a sonreír en la mañana,
volveré con lágrimas en los ojos,
mirar al cielo y dar las gracias.
Volveré a sentarme con los míos,
volveré a compartir mi alegría,
volveré pa contarte que he soñado,
colores nuevos y días claros.
Chambao, poquito a poco

Resumen

La restauración de espacios afectados por la minería de carbón a cielo abierto es una labor muy compleja. La creación de nuevos ecosistemas implica la activación de numerosos procesos naturales en el marco de la sucesión ecológica. Los procesos ecohidrológicos resultan claves en el funcionamiento de estos nuevos sistemas. Son ya conocidas las claves de la dinámica ecohidrológica de laderas mineras restauradas con régimen de escorrentía en regueros, caracterizadas por un avance limitado de la sucesión. Sin embargo, es muy escaso el conocimiento de laderas con régimen de escorrentía laminar y un mayor potencial de sucesión ecológica.

La presente Tesis Doctoral tiene como objetivo estudiar las interacciones entre vegetación e hidrología superficial y su relación con los patrones de sucesión ecológica, haciendo especial énfasis en el efecto que tiene en ello el volumen de escorrentía circulante por la ladera. La hipótesis de partida es que la interacción entre la escorrentía superficial y la vegetación es un fenómeno clave para explicar la estructura espacial y el funcionamiento de estos sistemas y, en particular, que la escorrentía es una fuerza directriz de la sucesión ecológica (agradación).

En un primer estudio a escala regional, se describe la estrecha interacción entre escorrentía y vegetación en laderas restauradas no reguerizadas, identificándose "unidades ecohidrológicas", que se diferencian por su composición florística y comportamiento hidrológico. La reguerización intensa aparece como un fenómeno degradativo en el que el control de la hidrología superficial es llevado a cabo por elementos abióticos. Por otro lado, se observa que la intensidad de las interrelaciones ecohidrológicas es menor en las laderas cubiertas de vegetación, por lo limitado de los procesos de escorrentía superficial.

Posteriormente se lleva a cabo una investigación experimental en tres laderas restauradas de manera similar, diferenciadas únicamente en el volumen de escorrentía circulante a causa de sus diferencias en la topografía de su cabecera. Estas laderas abarcan un gradiente de escorrentía desde la reguerización discontinua hasta el dominio de la escorrentía laminar débil.

En este contexto se analiza la influencia de la vegetación en la heterogeneidad espacial de los procesos hidrológicos superficiales identificando manchas de vegetación sumideros y fuentes de escorrentía. Además, se estudia la influencia del volumen de escorrentía superficial circulante por la ladera en la heterogeneidad hidrológica, concluyendo que éste controla la "diversidad hidrológica": cuando el volumen de escorrentía aumenta, la diversidad hidrológica disminuye, desarrollando comportamientos hidrológicos muy contrastados (fuentes extremas y sumideros).

Se explora también el efecto de la distribución espacial de la humedad edáfica -condicionada por las unidades ecohidrológicas- sobre la composición, estructura y dinámica de la vegetación, así como sobre los factores limitantes para la colonización vegetal, en función del volumen de escorrentía superficial circulante. Se analizan las relaciones funcionales entre UEH importadoras y exportadoras de escorrentía estudiando los efectos ecológicos de la interrupción de los flujos de escorrentía entre fuentes y sumideros en laderas con diferente volumen de escorrentía superficial. Con ello, se comprueba que el marco conceptual Trigger Transfer Reserve Pulse

(TTRP) -que explica las relaciones ecohidrológicas de ambientes semiáridos- se puede emplear en estas zonas restauradas. Este marco conceptual no había sido descrito en ambientes restaurados, donde puede tener una aplicación práctica para el desarrollo de estrategias de restauración adecuadas.

Los resultados obtenidos por esta Tesis Doctoral reflejan que cuando se dan volúmenes altos de escorrentía las manchas de vegetación son dependientes de la escorrentía circulante y el control del agua se ejerce de forma abiótica. Por otro lado, a volúmenes bajos de escorrentía, el control del agua se produce por fenómenos bióticos, apareciendo un matorral (*Genista scorpius*) capaz de modificar el patrón de redistribución de la escorrentía y de realizar interacciones bióticas (facilitación) que pueden propiciar un total cubrimiento de la ladera por parte de la vegetación. En conclusión, el volumen de escorrentía influye en el dominio del recurso hídrico, dirigiendo de esta forma el proceso sucesional de laderas restauradas.

La identificación de la escorrentía como una fuerza directriz de la sucesión en estos ambientes restaurados hace necesaria su incorporación en los proyectos de restauración. Surge de esta forma el concepto del "manejo experto de la escorrentía", que debe ser desarrollado y aplicado en los proyectos de minería para una correcta evolución de los ambientes restaurados.

Palabras clave: restauración de minería, escorrentía, ecohidrología, dinámica de la vegetación, manchas de vegetación, humedad edáfica, manejo experto de la escorrentía, sucesión ecológica.

Abstract

Coal mining restoration is a complex activity. The construction of these new environments includes the natural processes framed within ecological succession. Ecohydrological processes are key for the functioning of these new ecosystems. The key factors of the ecohydrological dynamics in rilled restored slopes, characterized by "arrested succession" are well known. However, there is a scarcity of studies on slopes under sheet flow processes where ecological succession could have a greater potential to achieve revegetation.

The main objective of this PhD thesis was to study the interaction between vegetation and overland flow and its relationships with ecological succession patterns, focusing on the effects of overland flow routing along the slope on these interrelationships. The main hypothesis is that the interaction between overland flow and vegetation is a key phenomenon that directs the spatial structure and the functioning of these systems, and in particular, that overland flow is a driving force for ecological succession (agradation).

In a first study at the regional scale, strong interactions between overland flow and vegetation in non-rilled restored slopes were described, identifying "ecohydrological units" that are characterized by a specific floristic composition and hydrological behaviour. Rilling processes were identified as degradation processes in which the control of hydrology is directed by abiotic elements of the ecosystem. Moreover, we identified that the strength of ecohydrological interactions is lower in vegetated slopes, due to the absence of overland flow processes.

Afterwards, an experiment was carried out in three restored slopes subjected to different volumes of overland flow routing along the slope caused by differences in up-slope structures. These slopes span a gradient of overland flow from discontinuous rilling to mild sheet flow.

In this context, the influence of vegetation on the spatial heterogeneity of hydrological processes was analyzed, identifying runoff sinks and sources. Additionally, we investigated the influence of overland flow routing along the slope on hydrological heterogeneity, concluding that overland flow controls "hydrological diversity": as the overland flow volume increases, hydrological diversity decreases, developing opposite hydrological behaviours (extreme sources and sinks).

We also explored the effect of the spatial distribution of soil moisture -conditioned by ecohydrological units- on vegetation composition, structure and dynamics, together with the limiting factors for vegetation colonization, related to the overland flow volume routing along the slope. The functional relationships between runoff sources and sinks were analyzed, considering the ecological effects of runoff exclusion along an overland flow gradient at the slope scale. In summary, we proved that the TTRP (Trigger Transfer Reserve Pulse) framework -which explains ecohydrological interactions in semiarid environments-, can be used in these restored environments. This conceptual framework has never been applied in restored environments, where it can have practical implications for the development of appropriate restoration strategies.

The results obtained in this PhD thesis highlight that when overland flow volumes are reached, vegetation patches are highly dependent on overland flow and the control of water resources is abiotic. However, under low overland flow volumes routing along the slopes, water control is biotic, and a shrub species appears (*Genista scorpius*) which is able to modify runoff redistribution patterns and whose biotic interactions (facilitation) can direct the system towards a vegetated slope. In conclusion, overland flow volume influences the control of water resources, hence, it directs successional processes in these restored slopes.

The identification of overland flow as a driving force in these restored environments calls for its incorporation in restoration projects. As a result, the concept of "runoff expert management" emerges, which should be delved into and applied to mining restoration projects in order to achieve a correct evolution of these restored environments.

Keywords: mining restoration, overland flow, ecohydrology, vegetation dynamics, vegetation patches, soil moisture, runoff expert management, ecological succession.

Indice

Capítulo 1. Introducción general

Capítulo 2. Área de Estudio

Capítulo 3. Ecohydrological interrelationships between vegetation patches and soil hydrological properties along a disturbance gradient: how sources and sinks of runoff determine a restoration threshold

Capítulo 4. Hydrological heterogeneity in mediterranean reclaimed slopes: runoff and sediment-yield at the patch and slope scales along a gradient of overland flow

Capítulo 5. Overland flow as a driving force for ecological succession: evidence from reclaimed slopes in semiarid environments

Capítulo 6. Ecohydrological interactions in artificial slopes along a disturbance gradient: insights for ecological restoration in drylands

Capítulo 1

A nuestra Ana

“Sé quien en verdad eres.
Descubre tus talentos y tu propósito en la vida.
Esto te llevará a hacer lo que amas
y porque haces las cosas con amor,
obtendrás lo que necesitas”

Erich Fromm

*Successful reconstruction of ecosystems, when it is achieved, has the reward that
it is the ultimate proof of our ecological understanding (Bradshaw, 1983)*

Capítulo 1

Introducción general

Minería, restauración y sucesión ecológica

La minería del carbón a cielo abierto es una actividad de gran impacto sobre todos los compartimentos del ecosistema (véase cuadro 1). Por ello, los proyectos de restauración minera deben afrontar el desafío de recuperar la funcionalidad de un sistema *cuasi* esquelético, donde la vegetación y los suelos no están bien desarrollados y las comunidades que aparecen representan estadios iniciales de la sucesión (Bradshaw 1983).

Los proyectos de restauración constituyen una oportunidad excelente para la comprensión de la dinámica funcional y estructural de los sistemas ecológicos (Aronson *et al*. 1993). De este modo, suponen un test práctico de las teorías sucesionales para desarrollar un sistema restaurado estable. Existen dos aspectos en los que los proyectos de restauración pueden centrarse: estructura y composición (acumulación de biomasa y especies) o funcionalidad (flujos de energía o ciclado de nutrientes; Walker *et al*. 2007).

En algunos casos, la restauración sin asistencia ("sucesión espontánea" o "restauración pasiva") obtiene grandes resultados (Prach y Hobbs 2008), pero este no es el caso de las explotaciones mineras en clima mediterráneo continental, donde se ha descrito cómo las coberturas de vegetación consecuencia de la colonización espontánea no son suficientes para asegurar la estabilidad de las laderas restauradas (Ninot *et al*. 2001). En ambientes semiáridos (como es el caso del área de estudio de esta Tesis Doctoral, la comarca de las Cuencas Mineras de Teruel), las escasas e irregulares precipitaciones dificultan los procesos de sucesión tras la explotación y hacen esencial el empleo de técnicas de restauración ("restauración técnica" o "restauración activa", véase cuadro 2). En este caso el objetivo de la aplicación de medidas de restauración es situar el sistema por debajo del umbral de degradación, de forma que pueda producirse una sucesión ecológica autogénica o espontánea (Whisenant *et al*. 1995).

En este contexto, la ecohidrología es una ciencia que se ha posicionado como una disciplina esencial para la restauración de zonas degradadas (Wilcox y Thurow 2006). Esto es especialmente reconocido en sistemas semiáridos, en los que se ha visto necesaria una aproximación interdisciplinaria que transcienda las fronteras entre ecología e hidrología a través de la disciplina híbrida de la ecohidrología (Wainwright *et al*. 1999a; Wilcox y Newman 2005; Newman *et al.* 2006; Turnbull *et al.* 2008). Wilcox y Thurow (2006) señalan que en las próximas décadas la investigación en ecohidrología debe dirigirse a las interrelaciones entre vegetación, degradación del suelo e hidrología a varias escalas, y centrarse en cómo estas interrelaciones se inician, manifiestan y perpetúan, lo que actualmente constituye una de las mayores deficiencias de la ciencia para el desarrollo de correctas estrategias de restauración.

La sucesión ecológica se define como el cambio en composición y estructura de las comunidades a lo largo del tiempo (Pickett y McDonnell 1989). Este concepto ha tenido gran importancia en el desarrollo de la ciencia de la Ecología, abarcando una gran variedad de procesos, y ofreciendo gran aplicabilidad para la gestión, conservación y restauración de ecosistemas (Walker y del Moral 2003; Walker *et al.* 2007).

Cuadro 1.1. Minería de carbón: Impactos ambientales y sociales y su regulación normativa en España

La minería del carbón a cielo abierto constituye una actividad de impacto de alta intensidad sobre una gran cantidad de variables ambientales. Estos impactos se pueden resumir de la siguiente manera (Nicolau y Ruiz 1986; Bustillo y López 1996):

1- Efectos sobre los suelos, la geología y la geomorfología. Se producen cambios drásticos en los suelos (por eliminación, alteración o substitución de estos por materiales estériles), y en la estructura y disposición de los materiales geológicos. Del mismo modo se producen modificaciones topográficas de gran envergadura (formación de escombreras, huecos de explotación, etc.).

2- Efectos sobre la circulación y calidad de las aguas. La creación de nuevas geoformas incide sobre la circulación superficial de las aguas, a través de la ruptura y alteración de las redes de drenaje, así como mediante la formación de nuevas superficies generadoras de escorrentía y sedimentos. La circulación de aguas subterráneas se ve modificada por los huecos de explotación (rompen la estructura de los acuíferos y provocan el descenso de los niveles freáticos). La calidad de las aguas superficiales y subterráneas se ve afectada por contaminación física (aumento de sólidos en suspensión) y química (por el contacto del agua con las capas de carbón en el hueco de explotación y la formación de procesos de lixiviación en escombreras con materiales estériles salinos y/o piríticos).

3- Efectos sobre la atmósfera. La calidad del aire se ve alterada por la emisión de polvo y la autocombustión de materiales carbonosos presentes en las escombreras. Del mismo modo, la minería es una fuente importante de ruidos (tránsito de maquinaria, sistemas de ventilación, transporte del mineral, etc.).

4- Efectos sobre las comunidades bióticas. La minería produce la destrucción completa de los ecosistemas en el área de explotación. Del mismo modo, los impactos de la explotación se extienden sobre las comunidades del entorno por la acción de diferentes agentes y actividades (polvo, ruido, sondeos, tránsito de maquinaria, vertidos, efecto barrera, etc.).

5- Usos del territorio. Se produce una ocupación directa de las extensiones que son explotadas, lo que supone la eliminación de los usos tradicionales (agricultura y ganadería), la alteración de la infraestructura básica (caminos vecinales, construcciones, cercas, parideras, pasos de ganado, abrevaderos, etc.) y la transformación de la estructura de la propiedad.

En el caso de España, el desarrollo de la minería a cielo abierto provocó una fuerte preocupación social en los años 80, que originó la elaboración de un marco legislativo específico. En 1982 se dictó una norma genérica a nivel estatal (el RD 2994/1982 de 15 de octubre) en relación a las actividades extractivas; dos años más tarde, se elaboró una norma específica para la minería del carbón (RD 1116/1984 de 20 de noviembre). Con la incorporación de España en la Comunidad Económica Europea se asume la obligación de cumplir la Directiva 85/337, transpuesta a la legislación nacional por medio del Real Decreto legislativo 1302/1986 de 28 de junio de Evaluación de Impacto Ambiental, que se desarrolla en el Real Decreto 1131/1988 de 30 de septiembre (ambos modificados actualmente por el Real Decreto Legislativo 1/2008 de 11 de enero). A estas normas de carácter estatal hay que sumar la numerosa legislación específica desarrollada por las Comunidades Autónomas. Todas estas normas introducen explícitamente la obligación de desarrollar estudios para la evaluación de los impactos previstos, conectados a los planes de restauración de los espacios afectados por la minería del carbón a cielo abierto.

En el caso de la minería en Teruel, un análisis a escala macro, indica que la minería en esta región ha generado entre el 0,7 y el 2% de la electricidad en España, provocando un pasivo ambiental de cerca de 1000 ha degradadas que contaminan las aguas y cauces naturales. Sin embargo, un análisis a escala micro, pone de manifiesto que las minas deficientemente restauradas afectan a diversos valores tangibles e intangibles del territorio, que son de especial importancia para los habitantes de la comarca. Por ello les asiste el derecho a la exigencia de una restauración de las zonas afectadas (Nicolau *et al.* 2010).

Aunque la distinción entre sucesión primaria y secundaria es conceptualmente clara (la sucesión primaria se da en sustratos nuevos sin legado biológico mientras que la secundaria es propia de sustratos edáficos con una historia previa y, por tanto, donde existe un legado biológico), a menudo ambas son difíciles de diferenciar sobre el terreno y suelen aparecer situaciones intermedias, por lo que algunos autores afirman que ambos conceptos constituyen más bien los extremos de un gradiente de disponibilidad de recursos y especies (Vitousek y Walker 1987). Esta situación intermedia -aunque desplazada hacia el extremo de la sucesión primaria- es el caso de las laderas procedentes de minería, en las que el sustrato edáfico procede de los suelos naturales arrancados, almacenados y extendidos (denominados "tierra vegetal"), deficientes generalmente de materia orgánica, actividad microbiológica y propágulos. Además, la situación de partida en estas restauraciones se caracteriza por una ausencia total de la vegetación.

Esta tesis se enmarca conceptualmente en el "nuevo paradigma" de la sucesión ecológica (Parker y Pickett 1997), en el que los ecosistemas se consideran entes abiertos cuya regulación depende del contexto en el que se encuentran y de eventos externos. El clima y las características abióticas no definen tanto al ecosistema, sino que establecen fronteras entre posibilidades y, dentro de éstas, los factores dependientes del contexto paisajístico como la disponibilidad de propágulos en las áreas cercanas y la incidencia de las perturbaciones son las que determinan finalmente la dinámica de los ecosistemas (Simpson 2002). A lo largo de este trabajo se utilizará el concepto de "fuerza directora" o "fuerza directriz" para referirse a los factores ambientales que dirigen el proceso sucesional.

La sucesión ecológica en ambientes mineros restaurados ha sido más estudiada para climas húmedos o sub-húmedos, donde no existen limitaciones en el recurso hídrico. En estos ambientes se han descrito como factores directores de la sucesión ecológica, la distancia a la fuente de propágulos y factores locales como las propiedades fisicoquímicas del suelo y el reciclaje de nutrientes (Novak y Konvicka 2006; Prach *et al*. 2007). En algunos casos, severas deficiencias en el suelo o toxicidad son los factores limitantes más relevantes (Bradshaw 1997).

En los sistemas restaurados de ambientes semiáridos, la dinámica de la vegetación se ve también afectada por los flujos de escorrentía y por la erosión (Nicolau 2002). El incipiente desarrollo de los suelos limita la infiltración y con ello favorece la circulación superficial de la escorrentía (Ward *et al*. 1982; Guebert y Gardner 2001). Este hecho se ve magnificado en algunos casos por aportes extra de escorrentía procedentes de otras áreas, ya sea por fallos o colapsos en las estructuras de drenaje de las restauraciones, o por deficiencias en el diseño topográfico de la restauración. En este sentido, diferentes investigaciones han identificado las entradas exógenas de escorrentía como una fuerza directriz de los sistemas restaurados hacia la degradación (Hancock y Willgoose 2004; Moreno-de las Heras *et al*. 2008).

Eco-hidrología de laderas semiáridas

Los ecosistemas semiáridos se caracterizan por una heterogeneidad espacial de los recursos edáficos y de la distribución de la vegetación. En estos ambientes se han descrito dos tipos de factores directores de esta heterogeneidad espacial: la erosión diferencial y la vegetación. En patrones dirigidos por la erosión diferencial -que da lugar a una heterogeneidad geomorfológica- (p. ej. Rostagno y del Valle 1988; Wainwright *et al*. 2002), la vegetación juega un papel pasivo, adaptándose a aquellas geoformas donde las condiciones son propicias para su desarrollo.

Cuadro 1.2. Explotación y restauración de la minería de carbón a cielo abierto

La minería del carbón a cielo abierto generalmente se desarrolla por el método de formación de cortas con transferencia entre paneles (ITGME 1996; Abril y Molina 1997). Este sistema se basa en la apertura de un hueco que progresa a lo largo de la superficie ocupada por el yacimiento, auto-rellenándose conforme el material es extraído.

El sistema de explotación mediante cortas con transferencia entre paneles consta de las siguientes fases:

1. Retirada de la vegetación y retirada y acopio de los suelos de la superficie de los terrenos que van a ser explotados.

2. Apertura del primer panel de explotación mediante la extracción de los materiales estériles que recubren las capas de carbón.

3. Vertido de los materiales extraídos en una superficie externa, conformando la denominada escombrera exterior.

4. Extracción y transporte del carbón a puntos de tratamiento y/o consumo.

5. Repetición de los pasos anteriores, con la apertura de un nuevo panel, a continuación del primero, siguiendo la dirección de las capas del yacimiento. Los nuevos materiales estériles extraídos se depositan en el interior del hueco dejado en el anterior panel, conformando una escombrera interior.

La configuración final (Fig. 1.a) queda constituida por una escombrera exterior de grandes dimensiones y una serie de escombreras interiores de menor tamaño, seguidas de un hueco final.

La restauración se suele acometer de forma simultánea a la explotación y generalmente abarca las siguientes intervenciones (Nicolau y Moreno-de las Heras 2006) (Fig. 1.b):

1) Realización del perfilado topográfico de los materiales vertidos. Esta acción determinará las formas finales del relieve: plataformas (superficies planas), laderas (taludes), pistas de comunicación, cunetas y balsas para el drenaje y el control de la escorrentía.

2) Extensión de los substratos (tierra vegetal o en su defecto estéril de mina de características físico-químicas óptimas) sobre las nuevas superficies y preparación de los mismos (labrado, enmiendas químicas y/o orgánicas).

3) Revegetación. Las operaciones de revegetación suelen ser siembra (por métodos manuales o hidrosiembra) de mezclas de semillas de especies herbáceas y/o la plantación de árboles y/o arbustos.

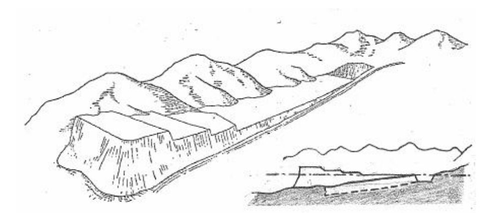


Figura 1.a. Configuración final ideal de una corta minera con transferencia por paneles. Tomado con modificaciones de Nicolau y Ruiz (1986).

Figura 1.b. Imagen que muestra de izquierda a derecha las tres fases de la restauración. Perfilado topográfico, extensión del substrato y revegetación.

Por el contrario, en patrones dirigidos por la vegetación, es ella la que inicia un proceso denominado "nucleación" mediante la obstrucción de flujos de: agua (Reid *et al.* 1999), sedimentos (Rostagno y del Valle 1988; Parsons *et al*. 1992; Sánchez y Puigdefabregas 1994) o nutrientes (Schlesinger *et al.* 1990). En este proceso se produce una retroalimentación positiva por la que la propia vegetación mejora las condiciones para el crecimiento de las plantas (Puigdefábregas *et al*. 1999).

En los últimos años, muchos estudios han subrayado la importancia de la redistribución de la escorrentía para la dinámica de ecosistemas áridos y semiáridos, con ejemplos en todo el mundo: en África (White 1970; Seghieri *et al*. 1997), América (Reid *et al.* 1999; Wilcox *et al.* 2003), Asia (White 1969), Australia (Dunkerley y Brown 1995; d´Herbès *et al*. 2001) y Europa (Calvo-Cases *et al.* 2003; Imeson y Prinsen 2004; Puigdefábregas 2005). El modelo de Activación-Transferencia-Reserva-Pulso (TTRP, Trigger-Transfer-Reserve-Pulse) propuesto por Ludwig *et al*. (1997; 2005) es un marco conceptual muy útil para explicar el papel de la redistribución espacial del agua en ecosistemas semiáridos y sus implicaciones para el desarrollo de la vegetación en los casos en que los patrones espaciales se dirigen por la acción de la vegetación (véase cuadro 3).

Esta redistribución del agua se ha interpretado como una adaptación de los ecosistemas semiáridos, puesto que los efectos de la interacción fuentes-sumideros (áreas productoras y receptoras de la escorrentía respectivamente) permiten incrementar la productividad si la comparamos con la obtenida por una distribución homogénea de la vegetación bajo la misma pluviometría (Aguiar y Sala 1994). Este mecanismo no sólo aumenta la eficiencia del uso del agua en el ecosistema (Noy-Meir 1973), sino que también mejora su estabilidad, porque le confiere resiliencia y capacidad adaptativa frente a perturbaciones (van de Koppel y Rietkerk 2004).

El patrón en fuentes y sumideros de laderas semiáridas fue descrito por vez primera por autores americanos que investigaron el matorral semiárido de algarrobo (*Prosopis* sp.) y de gobernadora (*Larrea tridentata*) del suroeste americano (Virginia y Jarrell 1983; Schlesinger *et al*. 1990). Sus investigaciones se centraron en describir la mejora de la fertilidad del suelo por parte de la vegetación, debida a una acumulación de nutrientes y a una mayor actividad biológica, desarrollando el concepto de "isla de fertilidad".

Un caso especial es el de los patrones en bandas de áreas semiáridas, que han sido descritos en el Sahel (Bromley *et al*. 1997; Seghieri y Galle 1999; Valentin *et al*. 1999), en Australia (Dunkerley y Brown 1995; Ludwig y Tongway 1995), o en el Norte de Méjico (Gutierrez y Hernandez 1996; Janeau *et al.* 1999). El hecho más reseñable de estos patrones es que en ellos la vegetación no sólo influye sobre los patrones espaciales de redistribución de escorrentía y sedimentos, sino que esta redistribución también produce una retroalimentación en la dinámica de las manchas de vegetación. Se ha demostrado que las bandas de vegetación de los "tiger bush" (Fig. 1.1) emigran ladera arriba (Chappell *et al*. 1999; 0.19-0.27 m año^{-1}) por la regeneración de la parte superior de las manchas de vegetación, mientras que se produce una senescencia de la parte inferior de la mancha por el agotamiento del agua disponible (Galle *et al.* 1999).

Cuadro 1.3. El marco conceptual Activación-Transferencia-Reserva-Pulso (Trigger-Transfer-Reserve-Pulse, TTRP) (Ludwig *et al*. 2005)

El marco conceptual TTRP explica las interacciones entre los procesos ecológicos e hidrológicos de sistemas áridos y semiáridos (Fig. 1.c) y muestra cuáles son los elementos que pueden verse más afectados por las perturbaciones (Ludwig *et al.* 1997; Ludwig y Tongway 2000). La precipitación (P) que cae sobre las manchas de vegetación y las zonas de suelo desnudo en las laderas puede ser infiltrada en el suelo (I), donde puede pasar a capas edáficas más profundas en función de las propiedades de conductividad hidráulica (K) (Fig. 1.d). Este agua puede percolar de forma profunda (DD) fuera del perfil del suelo. Si la intensidad de lluvia, junto con procesos que influyen en la incorporación del agua como la escorrentía cortical (Martinez-Meza y Whitford 1996), exceden la capacidad de infiltración del suelo, se produce la redistribución de la escorrentía (RO) que puede ser atrapada y retenida por una mancha cercana (RN), añadiéndose este agua a la reserva del suelo (DS). En zonas de suelo desnudo el agua del suelo se evapora (E), mientras que en las manchas de vegetación se producen fundamentalmente procesos de evapo-transpiración (ET) durante los pulsos de crecimiento de la vegetación. El agua almacenada en las capas del suelo (DS) también promueve la actividad biológica (B) para formar agregados del suelo y macroporosidad, aumentando así la infiltrabilidad, sobre todo en las manchas de vegetación. La escorrentía (RO) que circula sobre la superficie puede ser exportada fuera del sistema (D), llevando sedimentos en suspensión que pueden producir contaminación. Los componentes del marco conceptual TTRP se pueden tratar a distintas escalas temporales. Por ejemplo, una sola tormenta puede ser vista como un evento que activa la incorporación de agua en el paisaje en un breve periodo, o también se pueden tratar bajo este marco una serie de eventos en conjunto.

Este marco conceptual incluye también otros importantes procesos ecohidrológicos, como el proceso denominado "nucleación" (Puigdefábregas *et al.* 1999). Un pulso en el crecimiento de la vegetación produce más semillas y devuelve más hojarasca y nutrientes a la propia mancha de vegetación. Este pulso contribuye también de esta forma al mantenimiento de las manchas, incrementando su cobertura, lo que hace que sea a su vez más eficiente atrapando la escorrentía en el siguiente evento.

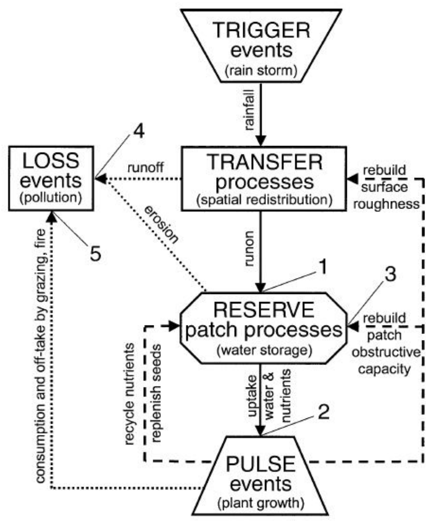


Figura 1.c. Marco conceptual Activación-Transferencia-Reserva-Pulso (Trigger-Transfer-Reserve-Pulse, TTRP). Los enlaces se representan con flechas continuas, las retroalimentaciones se representan con flechas discontinuas.

Otro aspecto muy útil de este marco conceptual es la interpretación de los efectos de las perturbaciones, como fuego o pastoreo, en el funcionamiento del ecosistema. Como se ha comentado antes, la cobertura (y la distribución espacial) de las manchas de vegetación en una ladera determinan el potencial del paisaje para obstruir el flujo del agua y con ello atrapar y retener el recurso (Tongway y Ludwig 1997). Si los paisajes semiáridos son intensivamente pastoreados o quemados a lo largo del tiempo, la cobertura de las manchas de vegetación se ve profundamente reducida, reduciendo también su efectividad para la obstrucción y aprovechamiento del recurso hídrico y por ello, incrementando la escorrentía y la erosión (Scanlan *et al.* 1996; Calvo-Cases *et al.* 2003).

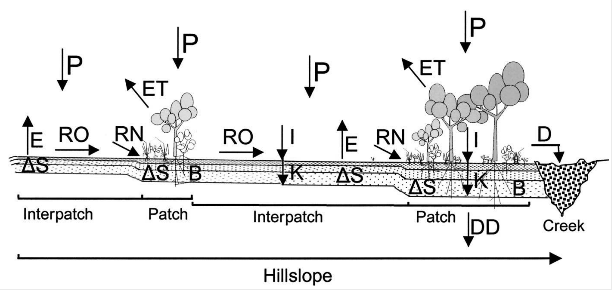


Figura 1.d. Diagrama ilustrando los procesos ecohidrológicos que ocurren en una ladera de pendiente moderada. Tomado de Ludwig (2005).

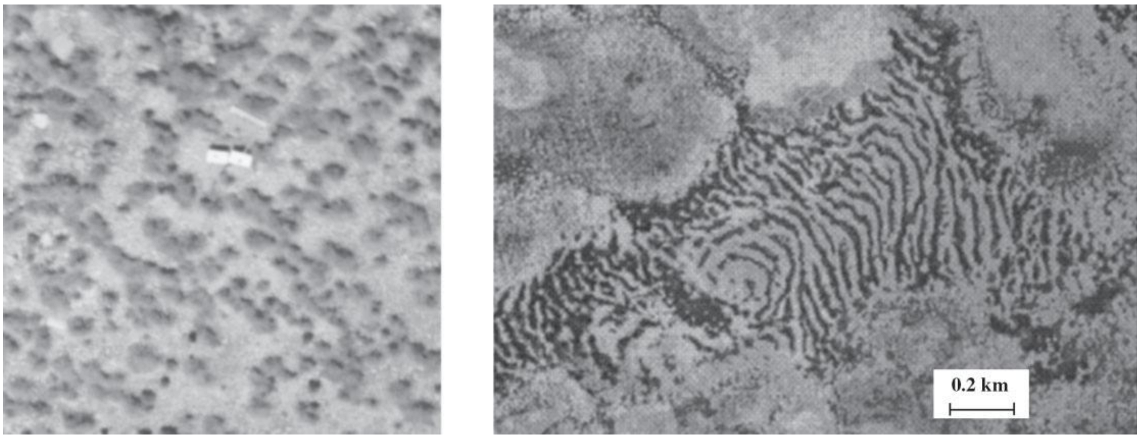


Figura 1.1. Fotografías aéreas mostrando la similaridad entre los patrones encontrados para *Stipa tenacissima* en Rambla Honda (fotografía de la izquierda), y el patrón "tiger bush" de Niger (fotografía derecha). En la fotografía de la izquierda la escala viene dada por la anchura de la parcela de escorrentía (2m), la dirección de la ladera es NE-SO, con una pendiente de 0.3%. Los puntos negros muestran las macollas, en gris oscuro aparece la parte senescente de las manchas con hojarasca y deposición de sedimentos y en gris claro aparecen las zonas de suelo desnudo con fragmentos de roca (blanco) en la superficie. En la fotografía de la derecha, la parte topográficamente más alta corresponde al centro-SE de la imagen. Las bandas de vegetación muestran la convexidad de la ladera y un límite marcado en la parte superior. Fotrografía de la derecha de Puigdefábregas, fotografía de la izquierda de J. Chadwick. Ambas imágenes han sido tomadas de Puigdefábregas (2005).

Asimismo, se han estudiado los procesos de "nucleación" en los que la vegetación produce modificaciones importantes sobre el suelo contribuyendo a su diferenciación de las áreas circundantes, lo que tiene fuertes implicaciones sobre la redistribución de escorrentía y sedimentos (Puigdefábregas 2005). De esta forma, son numerosos los estudios sobre los efectos de la vegetación en las propiedades del suelo y en el micro-ambiente bajo su dosel: radiación y temperatura (Domingo *et al*. 2000), partición del agua de lluvia (Domingo *et al*. 1994), efectos del sombreado (Domingo *et al*. 2001), reducción de la energía cinética de las gotas de agua (Wainwright *et al.* 1999b), estabilidad de agregados (Cerdà 1998), o fertilidad y estabilidad del suelo (Cammeraat y Imeson 1998; Barthes y Roose 2002).

Ecohidrología de laderas mediterráneas

En las laderas mediterráneas las zonas de suelo desnudo funcionan como áreas generadoras de escorrentía (fuentes), caracterizadas por bajas tasas de infiltración y una pobre estructura del suelo, mientras que las áreas cubiertas de vegetación funcionan como importadoras de escorrentía (sumideros), donde los contenidos en materia orgánica son mayores, favoreciendo la agregación del suelo y la actividad de la fauna edáfica, incrementando así la macro-porosidad y las tasas de infiltración (Sánchez y Puigdefábregas 1994; Cammeraat 2004). En efecto, la cobertura vegetal tiene una estructura espacial que interacciona de forma dinámica con los flujos de agua y sedimentos (Puigdefábregas *et al*. 1999). Como la ladera se compone de un mosaico de zonas de vegetación y de suelo desnudo, el flujo de escorrentía superficial es muy discontinuo debido a una infiltración "no-uniforme" (Cerdà 1998).

Calvo-Cases *et al*. (2003) han analizado la heterogeneidad espacial de la generación de escorrentía en laderas mediterráneas calizas observando que se comportan como un mosaico de áreas de generación e importación de escorrentía cuyo tamaño depende de las condiciones climatológicas, la morfología de la ladera, la litología, el desarrollo de los suelos y la intensidad de pastoreo a la que se ve sometida. En este sentido, Lavee *et al.* (1998) propusieron un modelo conceptual en el que tanto el tamaño de los claros que generan escorrentía como el ratio fuente/sumidero aumentan con la aridez (Fig. 1.2). Asimismo, señalan que las laderas mediterráneas pueden tener un comportamiento de generación de escorrentía característico de zonas áridas dependiendo del grado de interferencia humana mediante fuego, pastoreo o deforestación.

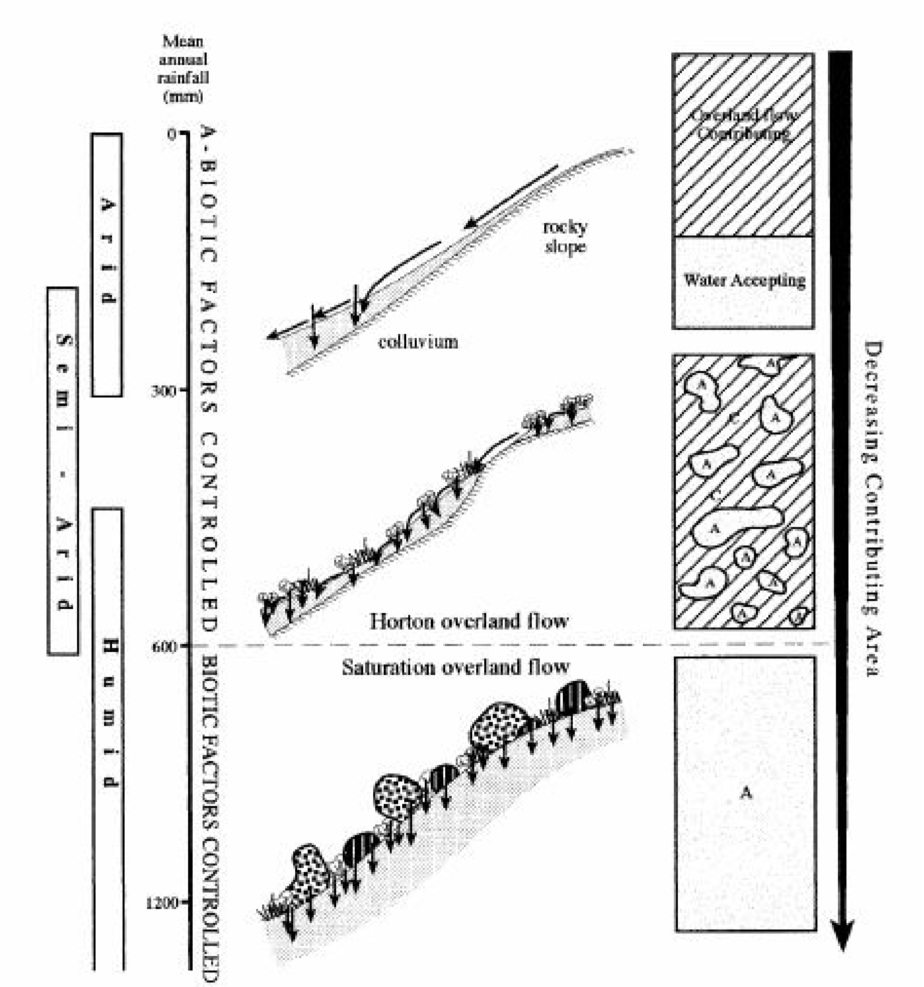


Figura 1.2. Redistribución del agua de lluvia bajo diferentes condiciones climáticas. (A: Área importadora de escorrentía, C: Zona exportadora de escorrentía). Tomado de Lavee *et al*. (1998).

El patrón en mosaico de áreas generadoras e importadoras de escorrentía asociado a la vegetación se ha observado también en espartales (*Stipa tenacissima*) de la Península Ibérica, donde se han descrito las implicaciones geomorfológicas de la redistribución de escorrentía (Sánchez y Puigdefábregas 1994; Puigdefábregas y Sánchez 1996). Cerdà (1997) encontró tres respuestas hidrológicas a la precipitación concluyendo que la distribución en manchas de *Stipa tenacissima* permitía la redistribución de escorrentía. Cammeraat e Imeson (1999) encontraron que en laderas con *Stipa tenacissima*, ésta se situaba en los bordes de las microterrazas construidas por la acumulación de sedimentos. Este proceso se ha encontrado también en tomillares y romerales en los que, una vez superada la capacidad de infiltración de la vegetación, la escorrentía se concentra e inicia la reguerización (Cammeraat 2004). Por útimo, se ha identificado un patrón espacial de la vegetación paralelo a las curvas de nivel (similar al patrón en bandas de la Fig. 1.1), con un efecto regulador sobre la generación de escorrentía y sedimentos a escala de ladera (Puigdefábregas *et al*. 1999).

Dinámica ecohidrológica de laderas semiáridas restauradas de la Cuenca Minera de Teruel

Las laderas (o taludes) construidas en las restauraciones constituyen un elemento (unidad paisajística o ecohidrológica) clave en el funcionamiento de los paisajes restaurados. Desde el punto de vista cuantitativo ocupan una superficie muy importante de los nuevos paisajes, y desde el punto de vista cualitativo su inestabilidad -por erosión hídrica superficial- puede convertirlos en la principal fuente de escorrentía y sedimentos, afectando negativamente sobre otros elementos de los paisajes restaurados (pistas, red de drenaje, humedales) y sobre los cauces naturales aguas abajo.

Las laderas derivadas de la restauración de la minería a cielo abierto se caracterizan, con frecuencia, por la importancia del flujo hídrico superficial que circula sobre ellas (Kapolka y Dollhopf 2001; Nicolau 2002). Los factores que favorecen la circulación superficial del agua, como ya hemos nombrado anteriormente, son dos principalmente: la baja permeabilidad de los sustratos y la entrada de escorrentía adicional (*run-on*) a la cabecera de las laderas procedente de fuentes externas como plataformas superiores, pistas o laderas situadas por encima. Por ello, estos ambientes podrían constituir una situación azonal en el esquema de Lavee *et al*. (1998), donde podremos encontrar laderas que se comportarían como laderas de sistemas semiáridos o laderas de sistemas húmedos dependiendo del volumen de escorrentía circulante por ellas (Fig. 1.2).

En las condiciones semiáridas (mediterráneo-continentales) de Teruel se ha descrito la colonización vegetal en las escombreras mineras y los factores de control de la siguiente manera (Moreno-de las Heras *et al*. 2008): En condiciones de sustratos ácidos, la colonización vegetal se ve muy limitada, siendo prácticamente inexistente debido a que la flora del entorno es de carácter basófilo. En la zona oriental de la provincia, bajo una mayor influencia mediterránea y donde los taludes fueron recubiertos con tierra vegetal aunque no revegetados, hay una activa colonización vegetal que es controlada por la distancia a las fuentes de propágulos. En la zona occidental, de carácter más continental, las herbáceas sembradas al inicio inhiben el establecimiento de las plantas del entorno. Sin embargo, cuando alguna perturbación (como pastoreo o presencia de hongos) abre huecos en la vegetación sembrada, hay colonización por parte de la vegetación autóctona. Finalmente, se identificado otro factor limitante más: el efecto erosivo de la escorrentía superficial que, al formar redes de regueros, condiciona notablemente el desarrollo de la vegetación.

Asimismo, el seguimiento de la vegetación y de sus causas de mortalidad en laderas reguerizadas ha permitido esclarecer que el principal efecto restrictivo que la erosión ejerce sobre la vegetación consiste en la disminución del contenido de humedad edáfica. Esta intensificación del déficit hídrico es propiciada por la erosión en regueros, que evacúa de forma eficiente la escorrentía y con ello, condiciona notablemente la colonización vegetal. En concreto, se ha comprobado que limita la emergencia de plántulas, su establecimiento y la producción de semillas (Espigares *et al*. *in press*).

Los regueros condicionan además la redistribución del agua en la ladera (concentrando la humedad del suelo en la base de los regueros), lo que afecta también a la distribución espacial de la vegetación. De esta forma, a partir de densidades de regueros de 0,60m/m^2 sólo una comunidad pauciespecífica de *Medicago sativa* (alfalfa) es capaz de desarrollarse, concentrándose los individuos a una distancia media entre el centro de los interregueros y de los regueros (localización en la que las plantas minimizan el estrés hídrico por un lado, y la perturbación mecánica por los flujos concentrados de escorrentía por otro). En estas condiciones, las pérdidas directas de recursos hídricos de las laderas en forma de escorrentía superficial representan más del 20% de la precipitación anual (Moreno-de las Heras 2009b). Cuando los niveles de cobertura vegetal son inferiores al 30% y las tasas de erosión en regueros superiores a 20 t ha^{-1} año^{-1} el sistema tiende a la degradación, mientras que coberturas del 50% y tasas de erosión en regueros inferiores a 5 t ha^{-1} año^{-1} permiten el desarrollo de comunidades vegetales (Moreno-de las Heras *et al*. 2009).

Por otro lado, las restricciones que la erosión impone al desarrollo de la vegetación dificultan considerablemente la incorporación de materia orgánica en el suelo. Los procesos de desarrollo y organización espacial de la estructura física y de la funcionalidad biológica del suelo se ven drásticamente limitados por la erosión. Entre ellos, el desarrollo de la estabilidad de agregados, el tamaño de las poblaciones microbianas y su actividad, así como la mineralización de la materia orgánica y el reciclado de los nutrientes del suelo (Moreno-de las Heras 2009a).

La presente Tesis Doctoral trata de ampliar el conocimiento de estos ecosistemas, abordando el estudio de laderas con un régimen hidrológico que abarca desde el dominio de los regueros discontinuos hasta la escorrentía laminar débil.

En estos sistemas se ha estudiado la interacción de la escorrentía circulante por las laderas restauradas y los procesos biológicos durante la sucesión ecológica dirigida hacia el cubrimiento vegetal de las laderas (que hemos denominado proceso de "agradación"). Si bien hay numerosa literatura científica que analiza la influencia de los procesos hidrológicos sobre la degradación de los sistemas (por sobrepastoreo, incendios u otras perturbaciones) (Davenport *et al.* 1998; Ludwig *et al*. 2000), apenas hay información sobre la influencia de la escorrentía superficial en la recuperación de los mismos. Este conocimiento es fundamental para dos ámbitos: de forma particular para los ecosistemas restaurados, pues puede contribuir decisivamente al desarrollo de protocolos adecuados de restauración, y de forma general para la teoría de la sucesión, pues aporta nociones novedosas sobre cómo evolucionan los procesos hidrológicos a lo largo de la sucesión.

Objetivos, hipótesis y escala de trabajo

La presente Tesis Doctoral tiene como objetivo general estudiar los procesos ecohidrológicos que operan en laderas restauradas de la minería de carbón a cielo abierto y su relación con los patrones fundamentales de sucesión ecológica, haciendo especial énfasis en el papel del volumen de escorrentía circulante por la ladera.

La hipótesis general de partida es que la interacción entre la escorrentía superficial y la vegetación es un fenómeno clave para explicar la estructura espacial y el funcionamiento de estos sistemas y que, en particular, la escorrentía es una fuerza directriz de la sucesión ecológica (agradación).

El análisis de las interacciones ecohidrológicas se ha realizado siguiendo el modelo conceptual TTRP. Los objetivos específicos y sus hipótesis a desarrollar dentro de la Tesis Doctoral se estructuran en los siguientes bloques:

1. Constatar la existencia de Unidades Ecohidrológicas (UEH) como expresión de la estrecha interacción de los procesos hidrológicos y de colonización vegetal y evaluar la intensidad de la interacción tanto en laderas naturales como restauradas. Se plantea como hipótesis que la interacción entre la escorrentía superficial y la estructura y composición de la vegetación debe ser significativamente superior en laderas restauradas que en laderas naturales (puesto que en ellas son más influyentes procesos bióticos, como competencia o facilitación). Asimismo, se plantea como hipótesis la existencia de UEHs que serían el resultado de la interacción entre la escorrentía y la vegetación.

2. Analizar la influencia de la vegetación en la heterogeneidad espacial de los procesos hidrológicos superficiales y la influencia en ella del volumen de escorrentía superficial circulante por la ladera. Como hipótesis fundamental se plantea la existencia de relaciones muy intensas entre la estructura y la composición de la vegetación y el comportamiento hidrológico de cada UEH, existiendo fundamentalmente dos comportamientos opuestos con respecto a la escorrentía: fuentes y sumideros. Asimismo, se plantea como hipótesis una interacción entre el volumen de escorrentía circulante por la ladera y el desarrollo de las unidades ecohidrológicas en el sentido que a menor volumen circulante disminuye el peso de las unidades "fuente" y aparecen los "sumideros profundos".

3. Explorar el efecto de la distribución espacial de la humedad edáfica -condicionada por las unidades ecohidrológicas- sobre la composición, estructura y dinámica de la vegetación, así como sobre los factores limitantes para la colonización vegetal en las unidades ecohidrológicas en función del volumen de escorrentía superficial circulante. Asimismo, identificar mecanismos de retroalimentación entre hidrología y vegetación. Se expone como hipótesis que en las laderas restauradas el principal factor limitante para el establecimiento y germinación de la vegetación es la humedad edáfica, que está controlada por el volumen de escorrentía superficial a escala de ladera, íntimamente relacionado con la distribución de las UEHs. Se plantea también como hipótesis que existe un umbral de volumen de escorrentía circulante que condiciona el peso de la vegetación o de la escorrentía superficial en su interacción mutua. Con niveles bajos de escorrentía la interacción ecohidrológica está más controlada por la actividad de la vegetación mientras que con niveles altos de escorrentía es la hidrología superficial la que tiene más peso en esta interacción recíproca.

4. Analizar las relaciones funcionales entre UEHs importadoras y exportadoras de escorrentía a través del estudio de los efectos ecológicos de la interrupción de los flujos de escorrentía entre fuentes y sumideros en laderas con diferente volumen de escorrentía superficial. Se plantea como hipótesis que la escorrentía es un factor director de la sucesión de laderas restauradas que contribuye a la redistribución del agua entre las diferentes UEHs, mejorando el estatus hídrico y el desarrollo de la vegetación en las UEHs que se comportan como sumideros de escorrentía. Asimismo, también se plantea como hipótesis la existencia de procesos de facilitación en la ladera sometida a menores volúmenes de escorrentía circulante.

Las cuatro interacciones ecohidrológicas fundamentales del modelo TTRP están explícita o implícitamente incorporadas en las hipótesis anteriores y se testan específicamente. Se trata de las siguientes:

- Las manchas de vegetación obstruyen la escorrentía y almacenan agua.

- El agua de escorrentía provoca un pulso en el crecimiento de la vegetación.

- Las manchas de vegetación mejoran la capacidad de infiltración del suelo.

- Las perturbaciones influyen en las salidas de la escorrentía a escala de ladera.

Para el desarrollo de los objetivos propuestos, la Tesis Doctoral se ha llevado a cabo a tres escalas de trabajo. Inicialmente se realizó un trabajo de prospección a escala regional, muestreando laderas no afectadas por minería y laderas restauradas del municipio de Utrillas (Teruel) con el objetivo de describir interacciones ecohidrológicas y su intensidad según el grado de perturbación de las laderas (objetivo 1). El resto de objetivos se realizaron a dos escalas: a escala de ladera y a escala de mancha (objetivos 2, 3, 4). Ambas escalas se exploraron en tres laderas del área experimental del Cerro del Moral que se seleccionaron según un gradiente de escorrentía con el objetivo de estudiar de forma pormenorizada las interacciones ecohidrológicas en laderas restauradas y su relación con la escorrentía superficial que por ellas circula.

Estructura de la memoria

Además del presente capítulo introductorio (**capítulo 1**), esta memoria de Tesis Doctoral está constituida por seis capítulos más (Fig. 1.3).

El **capítulo 2** comprende la descripción del área de estudio en el que se ha llevado a cabo el trabajo desarrollado en esta Tesis. En este capítulo se encuentra una descripción del clima, la vegetación y los suelos de la región en la que se ha realizado el primer trabajo de prospección. Asimismo, se encuentra una descripción pormenorizada de las tres laderas situadas en la Mina El Moral en las que se han realizado el resto de trabajos de la Tesis.

Los siguientes cuatro capítulos que componen el resto de la Tesis (capítulos 3 a 6) recogen los resultados de los trabajos realizados para el desarrollo de los objetivos propuestos. Estos capítulos se corresponden con artículos científicos aceptados o enviados a revistas internacionales especializadas. La presentación de estos capítulos se ha realizado manteniendo la estructura de las publicaciones, por lo que algunas secciones (como el área de estudio o metodología) podrán resultar redundantes.

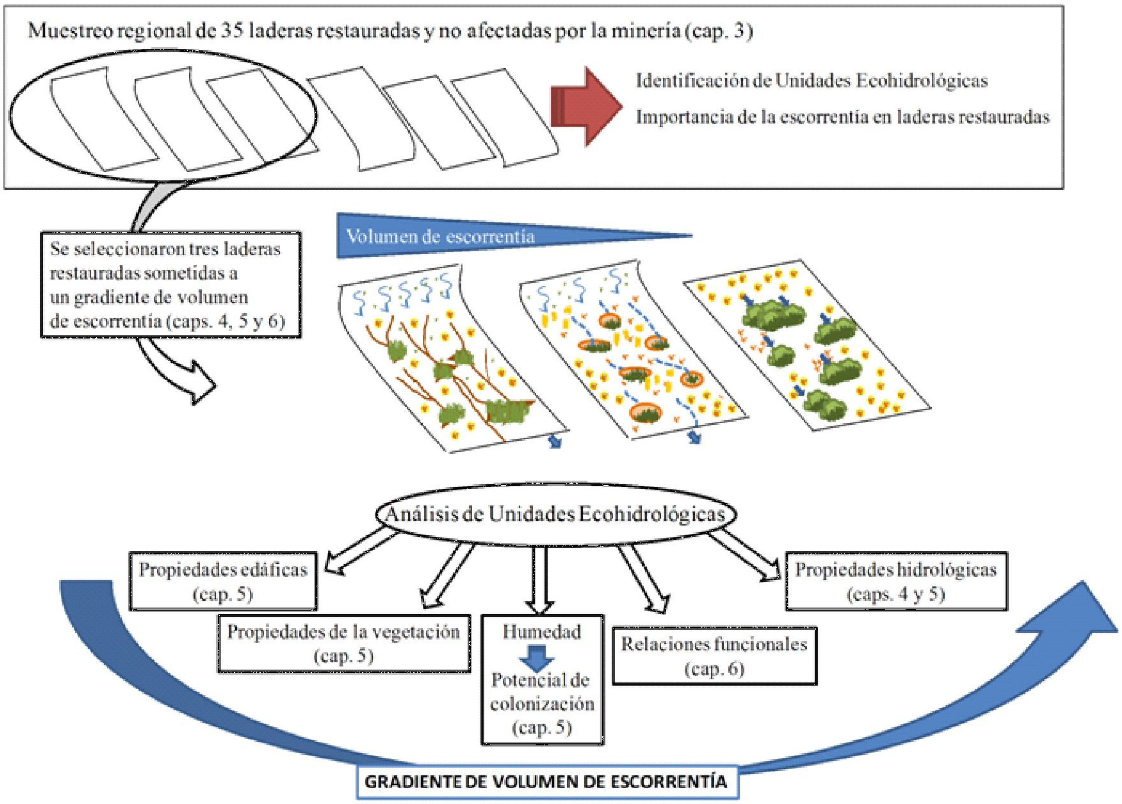


Figura. 1.3. Esquema general de los trabajos realizados en la Tesis Doctoral

El **capítulo 3** desarrolla el objetivo específico 1. Este trabajo se centra en la descripción de la intensidad de las interrelaciones vegetación-escorrentía en laderas naturales y en laderas restauradas. Asimismo, se realiza la identificación de unidades ecohidrológicas que surgen como resultado de estas interacciones. Para su desarrollo, se realizó un muestreo prospectivo en 35 laderas naturales y restauradas de comunidades vegetales y procesos hidrológicos.

El **capítulo 4** desarrolla el objetivo específico 2. Este apartado se centra en el análisis del comportamiento hidrológico de las unidades ecohidrológicas y en la influencia de la escorrentía circulante en la ladera para el desarrollo de unas u otras unidades. Para abordarlo, se realizó una monitorización en campo de la generación de escorrentía y sedimentos a escala de ladera y a escala de mancha (UEH) durante el periodo comprendido entre octubre de 2007 hasta diciembre de 2008. Asimismo, se cartografiaron las UEH presentes en las laderas.

El **capítulo 5** desarrolla el objetivo específico 3. Este trabajo estudia la dinámica de la humedad y de la vegetación de las UEH y sus relaciones con los volúmenes de escorrentía circulantes por la ladera. Para su estudio, se monitorizó la humedad edáfica en una red de sensores distribuidos en las UEH a distintas profundidades durante el periodo de estudio (octubre de 2007 a diciembre de 2008). Conjuntamente, se realizaron simulaciones de lluvia, análisis de la estructura de la vegetación y del banco de semillas y análisis de los requerimientos hídricos para la germinación de las especies más conspicuas de las laderas.

El **capítulo 6** desarrolla el objetivo específico 4. En este capítulo se estudian las relaciones funcionales entre las distintas UEHs exportadoras e importadoras de escorrentía. Para ello se evaluó si la redistribución de la escorrentía tiene influencia en el desarrollo de la vegetación y su relación con el volumen de escorrentía circulante por la ladera. Con este fin se realizó un experimento de exclusión de escorrentía en las manchas sumidero de las tres laderas del gradiente a lo largo del año hidrológico 2007-08. Se evaluó el impacto ecológico de la exclusión de escorrentía sobre la humedad edáfica, la producción de biomasa de la vegetación, el potencial hídrico de las especies leñosas y el banco de semillas.

A continuación, se presenta un capítulo en el que se realiza una síntesis general de la Tesis Doctoral (**capítulo 7**). Posteriormente, el **capítulo 8** presenta las conclusiones generales obtenidas. Estas conclusiones se presentan junto a recomendaciones para las prácticas de restauración de laderas artificiales en clima mediterráneo seco.

Referencias bibliográficas

Abril H. y Molina F. (1997). Restauraciones mineras de ENDESA en la cuenca de Teruel. In: Naturaleza Aragonesa, pp. 45-50.

Aguiar M.R. y Sala O.E. (1994). Competition, facilitation, seed distribution and the origin of patches in a patagonian steppe. Oikos, 70, 26-34.

Aronson J., Floret C., Le Floc'h E., Ovalle C. y Pontanier R. (1993). Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. I. A view from the south. Restoration Ecology, 1, 8-17.

Barthes B. y Roose E. (2002). Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. Catena, 47, 133-149.

Bradshaw A. (1997). Restoration of mined lands - using natural processes. Ecological Engineering, 8, 255-269.

Bradshaw A.D. (1983). The Reconstruction of Ecosystems: Presidential Address to the British Ecological Society, December 1982. Journal of Applied Ecology, 20, 1-17.

Bromley J., Brouwer J., Barker A.P., Gaze S.R. y Valentine C. (1997). The role of surface water redistribution in an area of patterned vegetation in a semi-arid environment, south-west Niger. Journal of Hydrology, 198, 1-29.

Bustillo M. y López C. (1996). Recursos minerales: tipología, prospección, evaluación, explotación, mineralurgia e impacto ambiental, Madrid.

Calvo-Cases A., Boix-Fayos C. y Imeson A.C. (2003). Runoff generation, sediment movement and soil water behaviour on calcareous (limestone) slopes of some Mediterranean environments in southeast Spain. Geomorphology, 50, 269-291.

Cammeraat E.L.H. (2004). Scale dependent thresholds in hydrological and erosion response of a semi-arid catchment in southeast Spain. Agriculture Ecosystems & Environment, 104, 317-332.

Cammeraat L.H. y Imeson A.C. (1998). Deriving indicators of soil degradation from soil aggregation studies in southeastern Spain and southern France. Geomorphology, 23, 307-321.

Cammeraat L.H. y Imeson A.C. (1999). The evolution and significance of soil-vegetation patterns following land abandonment and fire in Spain. Catena, 37, 107-127.

Cerdà A. (1997). The effect of patchy distribution of *Stipa tenacissima* L. on runoff and erosion. Journal of Arid Environments, 36, 37-51.

Cerdà A. (1998). Soil aggregate stability under different Mediterranean vegetation types. Catena, 32, 73-86.

Chappell A., Valentin C., Warren A., Noon P., Charlton M. y d'Herbes J.M. (1999). Testing the validity of upslope migration in banded vegetation from south-west Niger. Catena, 37, 217-229.

d´Herbès J.M., Valentin C., Tongway D.J. y Leprun J.C. (2001). Banded vegetation patterns and related structures. En: Banded vegetation patterning in arid and semiarid environments. Environmental processes and consecuences for management. (eds. Tongway DJ, Valentin C y Seghieri J).

Davenport D.W., Breshears D.D., Wilcox B.P. y Allen C.D. (1998). Viewpoint: Sustainability of pinon-juniper ecosystems - a unifying perspective of soil erosion thresholds. Journal of Range Management, 51, 231-240.

Domingo F., Puigdefabregas J., Moro M.J. y Bellot J. (1994). Role of vegetation cover in the biogeochemical balances of a small afforested catchment in southeastern Spain. Journal of Hydrology, 159, 275-289.

Domingo F., Villagarcia L., Boer M.M., Alados-Arboledas L. y Puigdefábregas J. (2001). Evaluating the long-term water balance of arid zone stream bed vegetation using evapotranspiration modelling and hillslope runoff measurements. Journal of Hydrology, 243, 17-30.

Domingo F., Villagarcía L., Brenner A.J. y Puigdefábregas J. (2000). Measuring and modelling the radiation balance of a heterogeneous shrubland. In. Blackwell Science Ltd, pp. 27-38.

Dunkerley D.L. y Brown K.J. (1995). Runoff and Runon Areas in a Patterned Chenopod Shrubland, Arid Western New-South-Wales, Australia - Characteristics and Origin. Journal of Arid Environments, 30, 41-55.

Espigares T., Heras M.M.-d.l. y Nicolau J.M. (in press). Performance of Vegetation in Reclaimed Slopes Affected by Soil Erosion. Restoration Ecology, doi: 10.1111/j.1526-100X.2009.00546.x.

Galle S., Ehrmann M. y Peugeot C. (1999). Water balance in a banded vegetation pattern: A case study of tiger bush in western Niger. Catena, 37, 197-216.

Guebert M.D. y Gardner T.W. (2001). Macropore flow on a reclaimed surface mine: infiltration and hillslope hydrology. Geomorphology, 39, 151-169.

Gutierrez J. y Hernandez, II (1996). Runoff and interrill erosion as affected by grass cover in a semi-arid rangeland of northern Mexico. Journal of Arid Environments, 34, 287-295.

Hancock G.R. y Willgoose G.R. (2004). An experimental and computer simulation study of erosion on a mine tailings dam wall. Earth Surface Processes and Landforms, 29, 457-475.

Imeson A.C. y Prinsen H.A.M. (2004). Vegetation patterns as biological indicators for identifying runoff and sediment source and sink areas for semi-arid landscapes in Spain. Agriculture Ecosystems & Environment, 104, 333-342.

ITGME (1996). Manual de restauración de terrenos y evaluaciones de impactos ambientales en minería. Instituto Tecnológico Geominero de España, Madrid.

Janeau J.L., Mauchamp A. y Tarin G. (1999). The soil surface characteristics of vegetation stripes in Northern Mexico and their influences on the system hydrodynamics - An experimental approach. Catena, 37, 165-173.

Kapolka N.M. y Dollhopf D.J. (2001). Effect of Slope Gradient and Plant Growth on Soil Loss on Reconstructed Steep Slopes. International Journal of Surface Mining, Reclamation and Environment, 15, 86-99.

Lavee H., Imeson A.C. y Sarah P. (1998). The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect. Land Degradation & Development, 9, 407-422.

Ludwig J. y Tongway D. (2000). Viewing rangelands as landscape systems. In: Rangeland desertification (eds. Arnalds O yArcher S). Kluwer Academic, Dordrecht, The Netherlands pp. 39-52.

Ludwig J., Tongway D., Freudenberger D., Noble J. y Hodgkinson K. (eds.) (1997). Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.

Ludwig J.A. y Tongway D.J. (1995). Spatial-organization of landscapes and its function in semiarid woodlands, Australia. Landscape Ecology, 10, 51-63.

Ludwig J.A., Wiens J.A. y Tongway D.J. (2000). A scaling rule for landscape patches and how it applies to conserving soil resources in savannas. Ecosystems, 3, 84-97.

Ludwig J.A., Wilcox B.P., Breshears D.D., Tongway D.J. y Imeson A.C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology, 86, 288-297.

Martinez-Meza E. y Whitford W.G. (1996). Stemflow, throughfall and channelization of stemflow by roots in three Chihuahuan desert shrubs. Journal of Arid Environments, 32, 271-287.

Moreno-de las Heras M. (2009a). Development of soil physical structure and biological functionality in mining spoils affected by soil erosion in a Mediterranean-Continental environment. Geoderma, 149, 249-256.

Moreno-de las Heras M. (2009b). Efectos ecológicos de la erosión en laderas derivadas de la restauración de la minería a cielo abierto. Tesis Doctoral. Departamento de Ecología. Universidad de Alcalá. Alcalá de Henares (Madrid).

Moreno-de las Heras M., Merino-Martín L. y Nicolau J.M. (2009). Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. Catena, 77, 39-47.

Moreno-de las Heras M., Nicolau J.M. y Espigares T. (2008). Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecological Engineering, 34, 168-178.

Newman B.D., Wilcox B.P., Archer S.R., Breshears D.D., Dahm C.N., Duffy C.J., McDowell N.G., Phillips F.M., Scanlon B.R. y Vivoni E.R. (2006). Ecohydrology of water-limited environments: A scientific vision. Water Resources Research, 42, 15.

Nicolau J.M. (2002). Runoff generation and routing on artificial slopes in a Mediterranean-continental environment: the Teruel coalfield, Spain. Hydrological Processes, 16, 631-647.

Nicolau J.M., Espigares T., Merino-Martin L., Moreno-de las Heras M., Nyssen S. y Pérez-Domingo S. (2010). Sobre la compatibilidad de la minería del carbón a cielo abierto con la sostenibilidad ambiental. El caso de las cuencas de Teruel. En: Plan director para la restauración edafopaisajística y recuperación ambiental de los espacios degradados por la minería del carbón en la comarca del Bierzo (eds. García Álvarez A y Cantó Portillo F). CIEMAT Madrid, pp. 237-261.

Nicolau J.M. y Moreno-de las Heras M. (2006). Restauración de ecosistemas alterador por movimiento de tierras. El caso de la minería de carbón a cielo abierto. En: Conservación del medio ambiente: revegetación, recuperación del suelo y empleo de residuos en el control de la erosión (eds. Bienes R y Marqués MJ). IMIDRA, Consejería de economía e innovación tecnológica Madrid, pp. 156-173.

Nicolau J.M. y Ruiz M. (1986). La minería del carbón a cielo abierto y sus repercusiones ambientales en el Bajo Aragón. In: Información Ambiental. Ministerio de obras públicas y urbanismo Madrid, pp. 49-60.

Ninot J.M., Herrero P., Ferre A. y Guardia R. (2001). Effects of reclamation measures on plant colonization on lignite waste in the eastern Pyrenees, Spain. Applied Vegetation Science, 4, 29-34.

Novak J. y Konvicka M. (2006). Proximity of valuable habitats affects succession patterns in abandoned quarries. Ecological Engineering, 26, 113-122.

Noy-Meir I. (1973). Desert Ecosystems: Environment and Producers. Annual Review of Ecology and Systematics, 4, 25-51.

Parker V.T. y Pickett S.T.A. (1997). Restoration as an ecosystem process: implications of the modern ecological paradigm. En: Restoration Ecology and Sustainable Development (eds. Urbanska KM, Webb NR y Edwards PJ). Cambridge University Press Cambridge, pp. 17-32.

Parsons A.J., Abrahams A.D. y Roger Simanton J. (1992). Microtopography and soil-surface materials on semi-arid piedmont hillslopes, southern Arizona. Journal of Arid Environments, 22, 107-115.

Pickett S.T.A. y McDonnell M.J. (1989). Changing perspectives in community dynamics - A theory of successional forces. Trends in Ecology & Evolution, 4, 241-245.

Prach K. y Hobbs R.J. (2008). Spontaneous succession versus technical reclamation in the restoration of disturbed sites. Restoration Ecology, 16, 363-366.

Prach K., Pysek P. y Jarosik V. (2007). Climate and pH as determinants of vegetation succession in Central European man-made habitats. Journal of Vegetation Science, 18, 701-710.

Puigdefábregas J. (2005). The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. Earth Surface Processes and Landforms, 30, 133-147.

Puigdefábregas J. y Sánchez G. (1996). Geomorphological implications of vegetation patchiness in semi-arid slopes. En: Advances in Hillslope Processes (eds. Anderson M y Brooks S). John Wiley & Sons Chichester, pp. 1027-1060.

Puigdefábregas J., Sole A., Gutierrez L., del Barrio G. y Boer M. (1999). Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in Southeast Spain. Earth-Science Reviews, 48, 39-70.

Reid K.D., Wilcox B.P., Breshears D.D. y MacDonald L. (1999). Runoff and Erosion in a Pinon-Juniper Woodland: Influence of Vegetation Patches. Soil Sci Soc Am J, 63, 1869-1879.

Rostagno C.M. y del Valle H.F. (1988). Mounds associated with shrubs in aridic soils of northeastern Patagonia: Characteristics and probable genesis. Catena, 15, 347-359.

Sánchez G. y Puigdefábregas J. (1994). Interactions of Plant-Growth and Sediment Movement on Slopes in a Semiarid Environment. Geomorphology, 9, 243-260.

Scanlan J., Pressland A. y Myles D. (1996). Run-Off and Soil Movement on Mid-Slopes in North-East Queensland [Australia] Grazed Woodlands. The Rangeland Journal, 18, 33-46.

Schlesinger W.H., Reynolds J.F., Cunningham G.L., Huenneke L.F., Jarrell W.M., Virginia R.A. y Whitford W.G. (1990). Biological Feedbacks in Global Desertification. Science, 247, 1043-1048.

Seghieri J. y Galle S. (1999). Run-on contribution to a Sahelian two-phase mosaic system: Soil water regime and vegetation life cycles. Acta Oecologica-International Journal of Ecology, 20, 209-217.

Seghieri J., Galle S., Rajot J.L. y Ehrmann M. (1997). Relationships between soil moisture and growth of herbaceous plants in a natural vegetation mosaic in Niger. Journal of Arid Environments, 36, 87-102.

Simpson T.B. (2002). An Open Approach to Ecosystem Change: Adopting a New Paradigm for Ecological Restoration and Land Management. Ecological Rest., 20, 190-194.

Tongway D.J. y Ludwig J.A. (1997). The conservation of water and nutrients within landscapes. En: Landscape ecology, function and management: principles from Australia's rangelands (eds. Ludwig JA, Tongway DJ, Freudenberger DO, Noble JC y Hodgkinson KC). CSIRO Publishing Melbourne, Australia, pp. 13-22.

Turnbull L., Wainwright J. y Brazier R.E. (2008). A conceptual framework for understanding semi-arid land degradation: ecohydrological interactions across multiple-space and time scales. Ecohydrology, 1, 23-34.

Valentin C., d'Herbes J.M. y Poesen J. (1999). Soil and water components of banded vegetation patterns. Catena, 37, 1-24.

van de Koppel J. y Rietkerk M. (2004). Spatial interactions and resilience in arid ecosystems. American Naturalist, 163, 113-121.

Virginia R.A. y Jarrell W.M. (1983). Soil properties in a mesquite-dominated sonoran desert ecosystem. Soil Science Society of America Journal, 47, 138-144.

Vitousek P.M. y Walker L.R. (1987). Colonization, succession and resource availability: ecosystem-level interactions. En: Colonization, succession and stability (eds. Crawley MJ, Edwards PJ y Gray AJ). Blackwell Scientific Oxford, UK, pp. 207-222.

Wainwright J., Mulligan M. y Thornes J.B. (1999a). Plants and water in drylands. In: Eco-hydrology: Plants and water in terrestrial and aquatic environments (eds. Baird AJ yWilby RL). Routledge, pp. 78-126.

Wainwright J., Parsons A.J. y Abrahams A.D. (1999b). Rainfall energy under creosotebush. Journal of Arid Environments, 43, 111-120.

Wainwright J., Parsons A.J., Schlesinger W.H. y Abrahams A.D. (2002). Hydrology-vegetation interactions in areas of discontinuous flow on a semi-arid bajada, Southern New Mexico. Journal of Arid Environments, 51, 319-338.

Walker L.R. y del Moral R. (2003). Primary succession and ecosystem rehabilitation, Cambridge.

Walker L.R., Walker J. y Hobbs R.J. (2007). Linking restoration and ecological succession. Springer.

Ward A.D., G. W.L. y E. P.R. (1982). Infiltration through reconstructed surface mine spoils and soils. Transactions of the ASAE, 26, 821-830, 832.

Whisenant S.G., Thurow T.L. y Maranz S.J. (1995). Initiating autogenic restoration on shallow semiarid sites. Restoration Ecology, 3, 61-67.

White L.P. (1969). Vegetation Arcs in Jordan. Journal of Ecology, 57, 461-464.

White L.P. (1970). Brousse Tigree Patterns in Southern Niger. Journal of Ecology, 58, 549-553.

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

Wilcox B.P. y Newman B.D. (2005). Ecohydrology of Semiarid Landscapes1. Ecology, 86, 275-276.

Wilcox B.P. y Thurow T.L. (2006). Emerging issues in rangeland ecohydrology: Vegetation change and the water cycle. Rangeland Ecology & Management, 59, 220-224.

Capítulo 2

EL MERCANTIL DE ARAGON

DIARIO INDEPENDIENTE

Defensor de la Agricultura, Industria y Comercio

Inauguración de las Minas y Ferrocarril de Utrilas

A Mariano

Es mucho lo que me ha dado esta parte de la provincia de Teruel, árido, montañoso y frío que guardaba su riqueza en las minas de carbón, de las que hemos vivido varias generaciones, hasta que dejó de ser rentable su explotación. Es por ello que me duele tanto ver como el paso del tiempo, la desidia, el abandono y la marginación han convertido en un valle de cenizas una parte de estas comarcas...

Un valle de cenizas, Francisco Rando Corella

Capítulo 2

Área de Estudio

Área experimental del Municipio de Utrillas (Teruel)

El área de estudio se encuentra enmarcada en la cuenca lignitífera de Teruel, cuenca del Río Martín, situada en la rama sureste del Sistema Ibérico (Fig. 2.1).

El régimen climático de la zona es Mediterráneo templado (*sensu* Papadakis 1966), con una precipitación anual de 480 mm, una evapotranspiración anual de 759 mm y una temperatura media anual de 14ºC (Peña *et al*. 2002).

El régimen de humedad se caracteriza por la concentración de episodios lluviosos en las estaciones de primavera y otoño, así como por la existencia de un periodo de sequía estival de dos a tres meses entre junio y septiembre, periodo en el que la evapotranspiración supera las aportaciones de agua por precipitación. Además, la zona se caracteriza por un corto periodo libre de heladas comprendido entre los meses de abril-mayo hasta septiembre (Peña *et al*. 2002, Fig. 2.2).

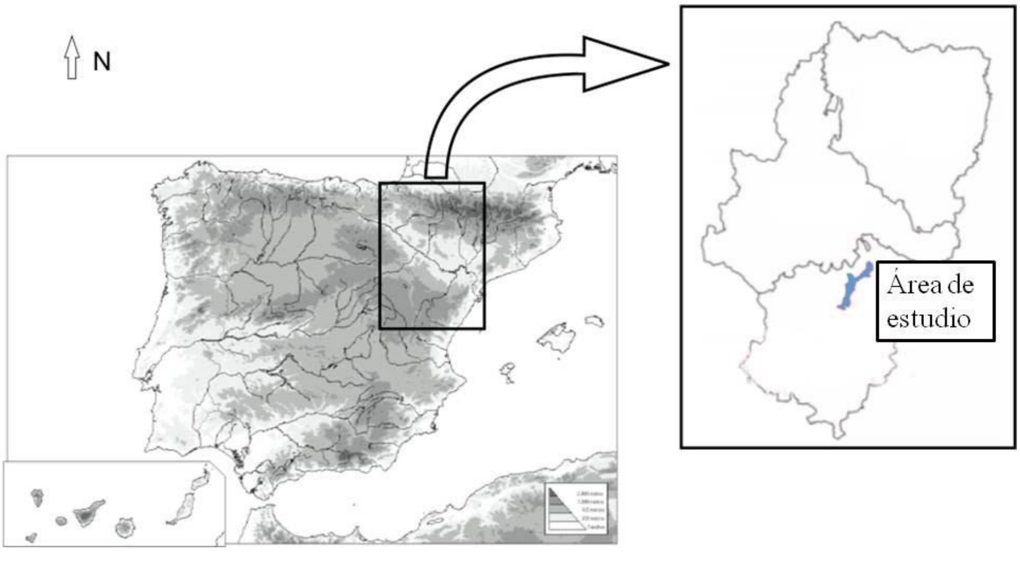


Fig. 2.1. Ubicación del área de estudio.

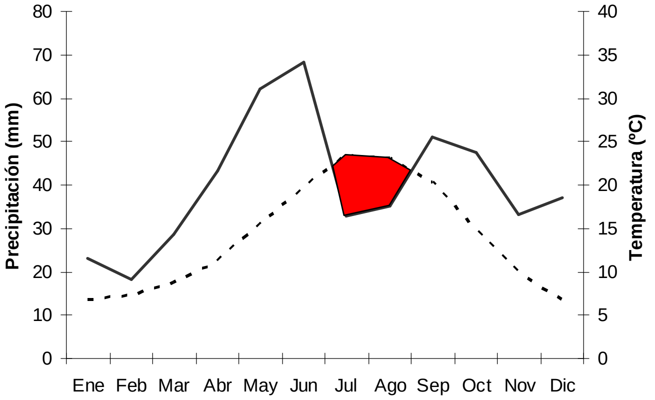


Fig. 2.2. Diagrama ombrotérmico (Climodiagrama de Walter-Gausen) del área de estudio (datos de la estación meteorológica de Montalbán).

El número medio anual de eventos pluviométricos en la zona es de 50 a 60 días, destacando la formación común de tormentas de fuerte intensidad, que pueden alcanzar los 100 mm en 24 h. El clima de la zona se caracteriza también por una alta variabilidad interanual, habiendo una dominancia de años secos que son interrumpidos por años húmedos de carácter excepcional (Peña *et al*. 2002).

La vegetación potencial varía de la dominancia de *Quercus ilex ballota* hacia la dominancia de *Q. faginea* (Rivas-Martinez 1987). El paisaje vegetal de la zona de estudio se compone de un mosaico de cultivos de cereal en las zonas planas, terrazas de cultivo abandonadas y vegetación discontinua en laderas dominadas por matorral de *Genista scorpius* y *Thymus vulgaris* (Montserrat 1990), así como de fracciones de espacios reforestados por *Pinus halepensis*. Los suelos de la región varían entre *Typic* y *Lithic Xerorthent* hasta *Calcic Xerochrept* (Arranz 2004), con pH neutro o moderadamente básico (7.3-8.6), generalmente pobres en materia orgánica (0.5-2.1%), de texturas franco arcillosas y pedregosidad elevada en algunas ocasiones (20-47% en peso).

El estudio regional que se realizó para abordar el objetivo 1 de esta Tesis Doctoral abarcó laderas restauradas de las minas de El Moral (40º47'50"N, 0º50'26"W), Yermegada (40º48'38.93"N, 0º52'11.13"W), y El Umbrión-Sabina (40º48'30.70"N, 0º52'55.84"W) y laderas no afectadas por actividades mineras del municipio de Utrillas (Fig. 2.3). La explotación y restauración de las minas fue ejecutada por la empresa M.F.U.S.A. (Minas y Ferrocarriles de Utrillas, S.A.) durante los años 1979-2001.

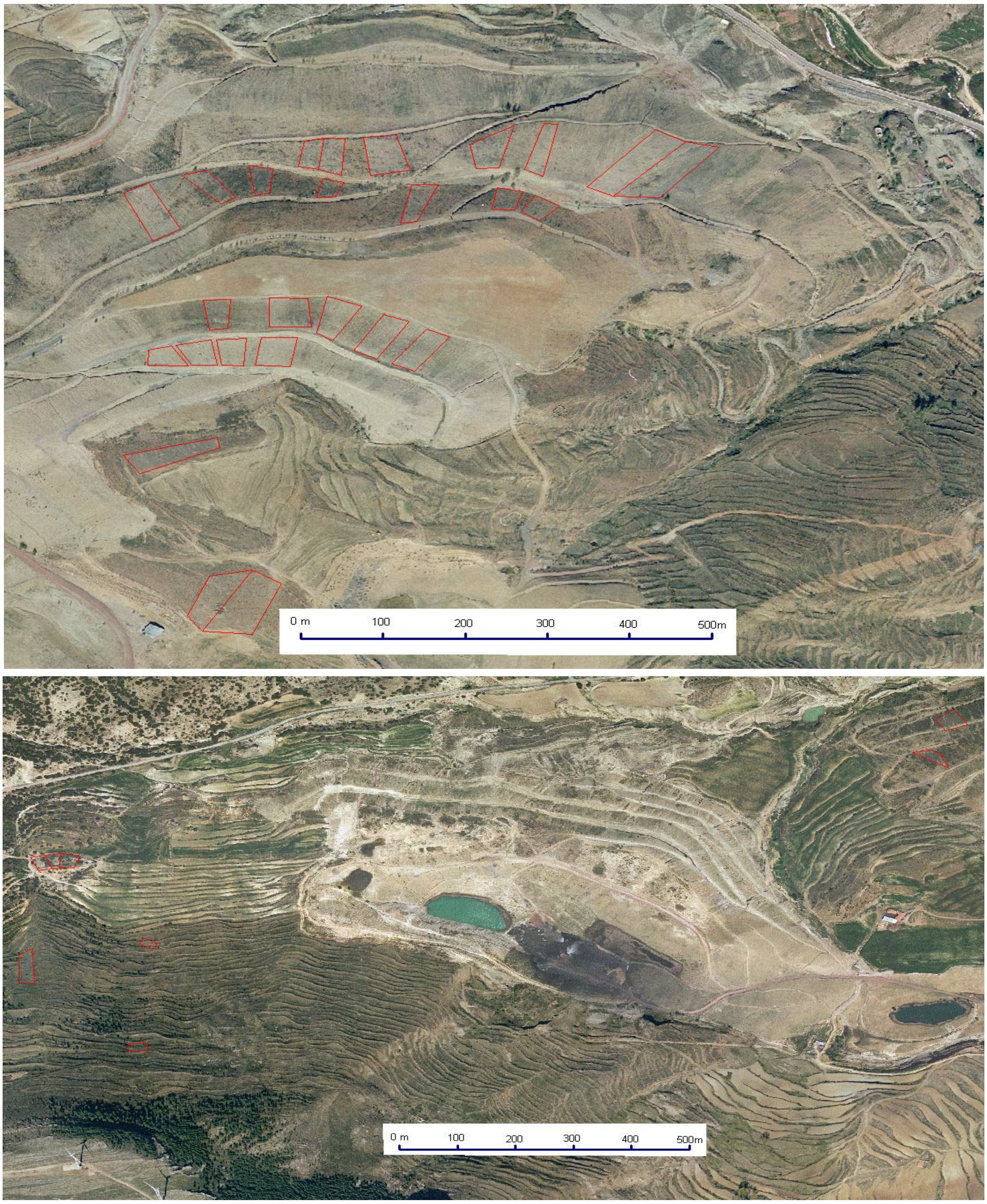


Figura 2.3. Laderas seleccionadas para la prospección a escala regional.

Descripción del modelo de restauración de Utrillas

La restauración llevada a cabo en Utrillas posee un enfoque innovador con respecto a las restauraciones realizadas habitualmente (Nicolau 2003). Esta restauración incorpora los procesos hidrológicos en el diseño de la geomorfología y se caracteriza por la compartimentación del espacio en cuencas con capacidad de regulación y por modelados de relieve suaves con pendientes en torno a 20º. El modelo Utrillas contrasta con las restauraciones clásicas de la minería de transferencia de modelo plataforma-talud-berma que buscan maximizar la superficie horizontal susceptible de ser explotada, lo que compromete la construcción de laderas de elevada inclinación (en torno a los 35º).

Una vez realizada la reconstrucción topográfica se procedió a la extensión de la tierra vegetal y un labrado perpendicular a las líneas de pendiente que originó la existencia de micro-terrazas. Como consecuencia de la extensión desigual de la tierra vegetal y la generación de viales en las partes altas de las laderas, se crearon lomas de mayor pendiente en la parte superior de las mismas.

Posteriormente se realizó una siembra inicial con una mezcla de especies herbáceas (gramíneas y leguminosas) que realizan una función de control de erosión, mejora del sustrato y estética. Las especies utilizadas en la siembra fueron: *Festuca rubra*, *Festuca arundinacea*, *Poa pratensis*, *Lolium perenne*, *Bromus catharticus*, *Medicago sativa*, *Onobrychis vicifolia* y *Melilotus officinalis*, todas estas especies son alóctonas excepto *Onobrychis viciifolia*, que procede de antiguos cultivos y ha sido naturalizada en la región (Montserrat 1990). Posteriormente en algunas laderas se plantó *Pinus halepensis*.

Desde la finalización de las prácticas de restauración en el año 2001, la superficie restaurada fue cedida a sus antiguos propietarios y al Ayuntamiento de Utrillas. Desde ese año hasta la actualidad, la zona restaurada ha estado sustentando una carga ganadera muy alta, lo que ha originado progresiva pérdida de cobertura vegetal.

Como resultado de la acción de estos factores bióticos y abióticos, la evolución de las laderas en esta zona tiene unos limitantes topográficos, climáticos, edáficos y de uso y gestión que dirigen la sucesión ecológica (Moreno-de las Heras 2009). Estos condicionantes son fundamentalmente:

- Estrés hídrico propiciado por la escasez e irregularidad de las precipitaciones junto con las altas tasas de evapotranspiración.

- Altas tasas de erosión propiciadas por la escasez de la vegetación y la frecuencia de eventos pluviales de fuerte intensidad.

- Corto periodo libre de heladas (abril-mayo hasta septiembre) durante el que se puede desarrollar la vegetación. Además este periodo libre de heladas se ve limitado por el periodo de sequía (dos a tres meses entre junio y septiembre).

- Sustrato estéril exclusivamente en algunas laderas, que impone condiciones edáficas más restrictivas para el desarrollo de la vegetación por su carácter arenoso y moderadamente ácido.

- Carga ganadera muy alta que disminuye la cubierta de vegetación porque dificulta el desarrollo de ésta a la vez que modifica las condiciones del suelo por compactación, produciendo un incremento en las tasas de erosión.

- Errores en el diseño topográfico que permiten la conexión de la escorrentía que circula por pistas de acceso y plataformas superiores con las laderas, provocando una acumulación de escorrentía que intensifica su efecto erosivo. Otros errores en el diseño de la topografía son la generación de lomas superiores, que constituyen superficies con una mayor pendiente que generan escorrentía que es exportada ladera abajo.

Área experimental de la mina El Moral (Utrillas)

La mina El Moral se encuentra ubicada en el término municipal de Utrillas, en el tercio más occidental de la cuenca lignitífera de la provincia de Teruel (40º47'24''N, 0º47'24''O). Esta mina se explotó desde la década de los 80 por la empresa Minas y Ferrocarril de Utrillas, S.A (M.F.U.S.A.).

La mina está situada a 1100 m s.n.m. en el Sistema Ibérico, sobre la vertiente norte de la Sierra de San Yust. La temperatura media anual es de 11º C. El clima es mediterráneo templado y el régimen de humedad es seco (Papadakis 1966). La precipitación anual es de 466 mm y el déficit hidrológico es de 292 mm desde junio hasta octubre (Nicolau 2002). El paisaje actual de la mina está compuesto por cultivos de cereal en las plataformas, terrazas de cultivo abandonadas y vegetación discontinua de matorrales *Genista scorpius*, *Santolina chamaecyparissus* y *Thymus vulgaris*.

En esta mina se seleccionaron tres laderas restauradas localizadas en la vertiente norte construidas durante los años 1988 y 1989 (Fig. 2.4). Los tratamientos de restauración empleados para estas tres laderas fueron los descritos anteriormente en el modelo de restauración de Utrillas.

Las tres laderas restauradas se seleccionaron según una particularidad en el diseño geomorfológico: la existencia de zonas de mayor pendiente (40º) y con escasa vegetación situadas en las cabeceras (Fig. 2.4). Estas zonas desprovistas de vegetación y con mayor pendiente actúan como áreas generadoras de escorrentía (áreas de contribución) que circula por la ladera. Por

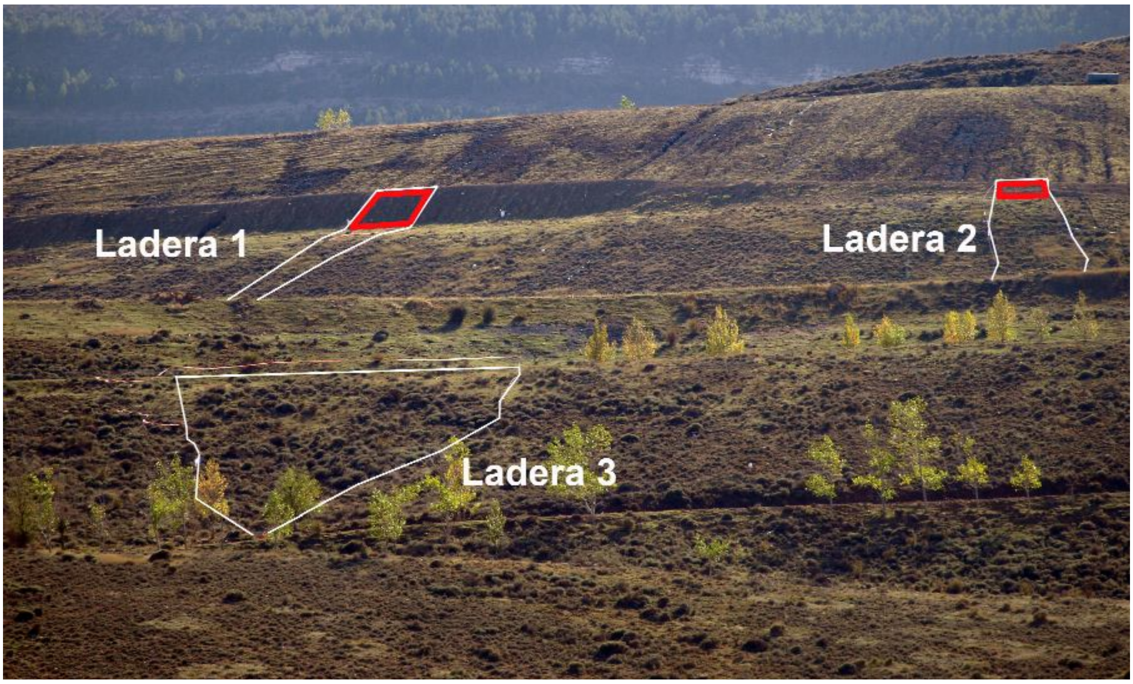


Fig. 2.4. Laderas seleccionadas en el gradiente de escorrentía; en rojo las áreas de contribución de escorrentía.

ello, se seleccionaron tres laderas con distinto tamaño de áreas de contribución, y con ello, distintos volúmenes de escorrentía circulante. Tras 20 años de sucesión ecológica, los procesos hidrológicos han dado lugar a diferencias en el desarrollo de la vegetación y la micro-geomorfología (Tabla 2.1).

A pesar de que las laderas estudiadas no constituyen réplicas en sentido estricto, pudiendo existir algunos factores que se escapen al estudio realizado en esta Tesis Doctoral, la similitud en su diseño y en los tratamientos de restauración que han recibido (salvo en el tamaño del área de contribución en las cabeceras) así como su cercanía física hace de estas laderas un escenario ambiental excepcional para el análisis de los efectos de la escorrentía sobre la sucesión ecológica de laderas restauradas en ambiente semiárido.

Tabla 2.1. Características generales de las laderas del estudio. Abreviaturas: N: Número de muestras; CE: Conductividad eléctrica; w/v: relación peso (suelo)/volumen (agua). Los valores son medias ± desviación estándar.

	N	Ladera 1		Ladera 2		Ladera 3	
Año de restauración		1988		1988		1987	
Topografía							
Área (m^2)		497.53		510.59		1474.30	
Pendiente (º)		20		20		20	
Área generadora de escorrentía (m^2)		50.36		22.67		0	
Orientación		Norte		Norte		Norte	
[1]*Características edáficas*							
Pedregosidad (%)	9	39.17 ± 4.54	a	40.50 ± 3.16	a	41.91 ± 3.36	a
Arenas (%)	9	44.75 ± 2.64	a	45.56 ± 2.68	a	43.95 ± 2.98	a
Limos (%)	9	28.68 ± 0.27	a	25.19 ± 1.07	a	29.75 ± 1.23	a
Arcillas (%)	9	26.57 ± 2.38	a	29.25 ± 1.61	a	26.30 ± 1.76	a
Textura	9	Franco arcillosa		Franco arcillosa		Franco arcillosa	
pH -H_2O; w/v: ½ -	9	8.38 ± 0.25	a	8.32 ± 0.15	a	8.01 ± 0.22	a
CE -w/v: ½- (dS m^{-1})	9	0.31 ± 0.06	a	0.31 ± 0.01	a	0.68 ± 0.42	a
Materia orgánica (%)	9	1.18 ± 0.25	a	1.55 ± 0.34	a	1.99 ± 0.37	a
$CaCO_3$ (%)	9	7.34 ± 0.44	a	6.17 ± 0.38	a	6.84 ± 0.28	a
[2]Densidad aparente (g cm^{-3})	27	1.48 ± 0.08	a	1.52 ± 0.03	a	1.43 ± 0.01	a
[3]*Cobertura*							
Suelo desnudo (%)	105	44.6 ± 3.1	a	32.3 ± 3.9	b	23.7 ± 2.8	b
Piedras (%)	105	25.5 ± 3.0	a	22.7 ± 1.6	a	21.1 ± 2.6	a
Hojarasca (%)	105	5.4 ± 1.8	a	1.1 ± 0.4	b	4.0 ± 1.9	ab
Vegetación (%)	105	24.4 ± 2.8	a	43.9 ± 4.1	b	51.2 ± 4.2	b
[3]*Vegetación*							
Riqueza de especies	105	3.8 ± 0.3	a	6.4 ± 0.4	b	9.3 ± 0.6	c
Índice Shannon	105	0.8 ± 0.1	a	1.2 ± 0.1	b	1.3 ± 0.1	b
Hidrología							
[4]Índice de erosión laminar	9	0.70 ± 0.09	a	0.59 ± 0.05	a	0.52 ± 0.06	a
[5]Densidad de regueros (m m^{-2})	3	0.58		0.00		0.00	
[6]Tasa de erosión en regueros[4] (t $ha^{-1}yr^{-1}$)	3	8.41		0.00		0.00	

[1] Medido en tres muestras compuestas (cada una formada por tres submuestras) de los 10 primeros centímetros en tres transectos distribuidos regularmente a lo largo de la ladera.

[2] Medido en 9 muestras inalteradas (3 cm de altura por 5 cm de diámetro) distribuidas al azar.

[3] Cobertura estimada visualmente durante la primavera de 2006 en 35 parcelas de 50 x 50 cm distribuidas regularmente.

[4] Medido por la relación: Cobertura de piedras / pedregosidad; según Moreno-del Heras (2008).

[5] Longitud lineal de reguero (m) por área (m^2).

[6] Medido a partir de las dimensiones de las redes de regueros según Morgan (1997).

Valores con letras distintas (a-c) difieren significativamente a α=0,05. Resultados de los análisis Kruskal-Wallis y U de Mann-Whitney test.

Determinaciones realizadas según la metodología descrita en MAPA (1994).

Referencias bibliográficas:

Arranz J.C. (2004). Propiedades, clasificación y evaluación de suelos mineros. Implicaciones sobre la ordenación y gestión de terrenos alterados por la minería. En: Escuela técnica superior de Ingenieros Agrónomos. Universidad Politécnica de Madrid Madrid, pp. 183-184.

MAPA (1994). Métodos Oficiales de Análisis. En: Dirección General de Política Alimentária. Ministerio de Agricultura, Pesca y Alimentación (ed. Alimentación SGd) Madrid, España.

Montserrat P. (1990). Vegetación. En: Mapa forestal de españa: Daroca (ed. de la Torre R). Ministerio de Agricultura, Pesca y Alimentación, ICONA Madrid, pp. 47-68.

Moreno-de las Heras M. (2009). Efectos ecológicos de la erosión en laderas derivadas de la restauración de la minería a cielo abierto. Tesis Doctoral. Departamento de Ecología. Universidad de Alcalá, Alcalá de Henares, Madrid.

Nicolau J.M. (2002). Runoff generation and routing on artificial slopes in a Mediterranean-continental environment: the Teruel coalfield, Spain. Hydrol Process, 16, 631-647.

Nicolau J.M. (2003). Trends in relief design and construction in opencast mining reclamation. Land Degrad Dev, 14, 215-226.

Papadakis J. (1966). Climates of the world and their agricultural potentialities, Buenos Aires.

Peña J.L., Cuadrat J.M. y Sanchez M. (eds.) (2002). El clima de la provincia de Teruel. Instituto de Estudios Turolenses (CSIC), Teruel.

Rivas-Martinez S. (1987). Memoria del mapa de series de vegetacion de España. Ministerio de Agricultura, Pesca y Alimentación, Madrid.

Capítulo 3

A Camilo y Paula

Soñé otro mundo, tan lejos y tan cerca
soñé otro viaje, cuatro caminos, cinco destinos
soñé la risa, soñé la ilusión
soñé otro mundo...

Soñé otro mundo, Manu Chao

Capítulo 3

Interrelaciones ecohidrológicas entre manchas de vegetación y propiedades hidrológicas del suelo en un gradiente de perturbación: cómo fuentes y sumideros determinan un umbral para la restauración

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Luis Merino-Martín; David D. Breshears; Mariano Moreno-de las Heras; Juan Camilo Villegas; Silvia Pérez-Domingo; Tíscar Espigares y José Manuel Nicolau. Aceptado. Ecohydrological interrelationships between vegetation patches and soil hydrological properties along a disturbance gradient: how sources and sinks of runoff determine a restoration threshold. Enviado a *Restoration Ecology*.

Resumen

La vegetación, el suelo y la hidrología en ambientes áridos presentan frecuentemente intensas interrelaciones en las que la vegetación influye y es influida por la escorrentía, particularmente en laderas de pendiente moderada. Estas interrelaciones tienen importantes implicaciones para la restauración ecológica de ambientes perturbados, como son aquellos ambientes restaurados de la minería, sin embargo, son escasos los estudios realizados desde ambas perspectivas evaluando específicamente cómo la intensidad de las interacciones ecohidrológicas varía entre laderas naturales y degradadas. Por ello, en este trabajo se evaluaron las interrelaciones entre la vegetación y las propiedades hidrológicas del suelo, estudiando los patrones de la estructura de la comunidad y las características hidrológicas del suelo relacionadas con la escorrentía para laderas naturales y restauradas de la minería empleando un análisis TWINSPAN (Two Way Indicator Species Analysis), un Análisis de Componentes Principales (ACP), una clasificación CLUSTER, un análisis PERMANOVA anidado (Permutational Multivariate Analysis of Variance) y un análisis de Redundancia (RDA). A escala de mancha, se identificaron ocho unidades ecohidrológicas a lo largo de un continuo desde fuentes a sumideros de escorrentía a partir de las interrelaciones entre las comunidades vegetales y las propiedades hidrológicas asociadas con la escorrentía. Igualmente, a escala de ladera, que incluía unidades ecohidrológicas de diferente tipología, se encontró correspondencia entre la abundancia de fuentes y sumideros de escorrentía y las propiedades hidrológicas y de la vegetación. La intensidad relativa de las interacciones ecohidrológicas en los mosaicos de las laderas disminuyó según se reducía la intensidad de la perturbación excepto para laderas reguerizadas, probablemente porque los regueros aíslan partes de la ladera entre sí. Nuestros resultados, en general, destacan que las interacciones ecohidrológicas están relacionadas con el grado de perturbación, y en particular, que la reguerización altera las interrelaciones ecohidrológicas, impidiendo de esta forma el desarrollo de una restauración efectiva.

Palabras clave: ecohidrología, interrelaciones ecohidrológicas, minería, restauración, ambientes secos, ladera, perturbación.

Ecohydrological Interrelationships Between Vegetation Patches and Soil Hydrological Properties Along a Disturbance Gradient: How Sources and Sinks of Runoff Determine a Restoration Threshold

Luis Merino-Martín[1], David D. Breshears[2,3], Mariano Moreno-de las Heras[4], Juan Camilo Villegas[2,5], Silvia Pérez-Domingo[1], Tíscar Espigares[1] and José-Manuel Nicolau[6].

[1]Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, Madrid, España.
[2]School of Natural Resources and the Environment, The University of Arizona. Tucson, Arizona, USA.
[3]Department of Ecology and Evolutionary Biology, The University of Arizona. Tucson, Arizona, USA.
[4]Faculty of Engineering and Built Environment, University of Newcastle, Callaghan, Australia.
[5]Grupo GIGA. Facultad de Ingeniería. Universidad de Antioquia. Medellín, Colombia.
[6]Departamento de Agricultura y Economía Agraria, Escuela Politécnica, Universidad de Zaragoza, Huesca, España.

Abstract

Vegetation, soil, and hydrology in drylands often collectively exhibit strong ecohydrological interrelationships in which vegetation both influences and is influenced by runoff, particularly on sites with more gradual slopes. These two-way relationships have important implications for ecological restoration of disturbed sites, such as those being reclaimed following mining, yet studies from both ecological and hydrological perspectives specifically evaluating how the strength of ecohydrological interrelationships varies for a range of natural and restored conditions are still missing. We assessed two-way relationships between vegetation and soil hydrological properties by evaluating patterns of both plant community structure and soil hydrological characteristics related to runoff for sites that were natural or restored following mining, using Two Way Indicator Species Analysis (TWINSPAN), a Principal Component Analysis (PCA), CLUSTER classification, nested permutational multivariate analysis of variance (PERMANOVA), and Redundancy Analysis (RDA). At the plot scale, we identified eight ecohydrological units based on interrelationships between vegetation communities and hydrological properties associated with runoff along a progression from source to sink patch types. Similarly, at the hillslope scale, which included patches of different types, we found a correspondence between the proportions of source and sink patches and both vegetation community and hydrological properties. The relative strength of ecohydrological interrelationships in hillslope mosaics decreased with decreasing disturbance except for rilled hillslopes, likely because parts of the hillslope become isolated from the others. Our results highlight in general how ecohydrological interrelationships are related with degree of disturbance, and in particular, how rilling alters ecohydrological interrelationships, thereby precluding effective restoration.

Key words: ecohydrology, ecohydrological interrelationships, mining, restoration, drylands, hillslope, disturbance.

Introduction

Interrelationships between ecology and hydrology are increasingly recognized as central to environmental processes in drylands (Ludwig *et al.* 1997; Rodriguez-Iturbe & Porporato 2004; Wilcox & Thurow 2006). Most importantly, interactions between ecological and hydrological processes can generate interrelationships that determine patterns and drive ecosystem processes. One such fundamental ecohydrological interrelationship in drylands is that between vegetation and the redistribution of runoff (Tongway *et al.* 2001; Ludwig *et al.* 2005; McDonald *et al.* 2009). Many studies document relevant aspects of this interrelationships, such as the effects of vegetation changes on runoff or erosion processes (Schlesinger *et al.* 1990; Abrahams *et al.* 1995; Abrahams *et al.* 2003; Michaelides *et al.* 2009; Moreno-de las Heras *et al.* 2009), the general dynamics of soil-water-interactions (Gutierrez-Jurado *et al.* 2006; Zea-Cabrera *et al.* 2006) and their spatial patterns (Puigdefabregas 2005; Bautista *et al.* 2007; Mayor *et al.* 2008), or scale-dependencies in runoff and erosion (Bergkamp 1998; Wilcox *et al.* 2003; Moreno-de las Heras *et al.* 2010; Urgeghe *et al.* 2010). The importance of such feedbacks is evident in modeling studies (Tongway *et al.* 2001; Caylor *et al.* 2006; Caylor *et al.* 2009). Some studies focuses on how disturbance affects runoff or erosion rates (Castillo *et al.* 1997; Wilcox *et al.* 2003), but less clear, however, are the nature and the degree to which ecohydrological interrelationships change with disturbance.

How ecohydrological processes and associated interrelationships change following disturbance is directly relevant to challenges associated with restoration of degraded landscapes (Eamus *et al.* 2006; Wilcox & Thurow 2006). Restoration of drylands focuses not only on reestablishing patterns of vegetation cover, but also on restarting key ecosystem processes (Aronson *et al.* 1993; Suding & Hobbs 2009). Among the most extreme restoration challenges are reclamation at mine sites. Mining reclaimed terrains are characterized by a rudimentary structure with undeveloped to poorly developed soils and vegetation (Bradshaw 1983). Restoring key processes in such extremely disturbed sites poses major challenges.

Recent successful mine restoration projects indicate that careful application of technical reclamation procedures and techniques can produce structurally and functionally diverse systems (Koch & Hobbs 2007). However, mining reclamation projects have historically failed in the application of comprehensive conceptual frameworks, their general understanding of reference ecosystems, long-term planning, and consideration of contingencies (Nicolau & Moreno-de las Heras 2005). Generally, rehabilitation of ecohydrological function in mining terrains starts with mechanical treatments such as furrowing or creation of micro catchments (Manu *et al.* 2000), but these treatments are usually inefficient in the long term because of increased erosive potential from the land surface (Macdonald & Melville 1999). Furthermore, these problems have been addressed by the incorporation of keystone species that can be sustained in the long term (Whisenant *et al.* 1995). Consequently, the application of mechanistic ecohydrological criteria that explicitly considers the interrelationships between vegetation and hydrology is critical to achieve primary management objectives such as optimization of water yields and restoration of degraded areas (Wilcox & Thurow 2006).

To assess how ecohydrological interrelationships vary with disturbance, we evaluated patterns of both plant community structure and soil hydrological characteristics related to runoff in different sites that cover a broad spectrum of disturbance scenarios (natural areas subjected or not to grazing pressure and restored areas following mining with different levels of success). We specifically considered individual vegetation patches at a plot scale and how they related to patterns and responses at the hillslope scale. Our main objectives were: (1) to evaluate ecohydrological interrelationships at the patch scale, considering the interactions between vegetation communities and hydrological processes in both restored and natural slopes; (2) to explicitly evaluate two-way ecohydrological interrelationships between hydrological processes and vegetation structure at the hillslope scale; (3) to compare the strength of ecohydrological interrelationships among restored and natural hillslopes. We hypothesized (a) that individual patches defined by either vegetation community type or hydrological properties related to runoff would be interrelated; (b) that this interrelationship is evident at larger scales that include a mosaic of individual patches and is based on the relative proportions of source and sink patches; and (c) that the relative strength of ecohydrological interrelationships decreases inversely with disturbance from restored to natural mosaics.

Study Site and Methods

Study Area

Our study site was located within the *Utrillas* coalfield (~1100 m above sea level) in the Iberian Mountain Chain in Spain and encompassed a group of hillslopes that spanned a broad range of restoration stages. We selected 25 hillslopes that had been restored and revegetated following construction located in three different reclaimed mine spoil banks (*El Moral* at 40º47'50"N, 0º50'26"W, *Yermegada* at 40º48'38.93"N, 0º52'11.13"W, and *El Umbrión-Sabina* at 40º48'30.70"N, 0º52'55.84"W) and 10 natural slopes unaffected by mining activities (40º48'29.39"N, 0º52'23.18"W). The climate in the area is Mediterranean-Continental type with a mean annual temperature of 14 ºC (ranging from a minimum mean daily temperature of 6.8 ºC in December and a maximum mean daily temperature 23.5 ºC in July), with air frost period between October and April. The local moisture regime can be classified as dry Mediterranean (Papadakis 1966) with mean annual precipitation of 480 mm (mainly concentrated in spring and autumn) and potential evapotranspiration of 759 mm, yielding a hydrological deficit of 292 mm running from June to October. The mean number of annual rainfall events in the area is ~50, with some convective rainstorms occurring especially in summer, characterized by high rainfall intensities of up to 100 mm in 24 hours (Peña *et al.* 2002).

The constructed hillslopes were built between 1985-1989 by the *Minas y Ferrocarril de Utrillas S.A.* mining company to have slopes between 20º and 30º and a layer of 100-250 cm of overburden substratum (from the *Escucha* and *Utrillas* cretacic formations of *Albian* age) spread over the spoil bank; the substratum is clay-loam. Revegetation of the slopes was implemented by cross-slope sowing with a mixture of perennial grasses (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis* and *Lolium perenne*) and leguminous herbs (*Medicago sativa* and *Onobrychis viciifolia*). Although the hillslopes were restored using the same general procedures, they differed in their subsequent evolution (ie. rilling, vegetation development), apparently due to differences in topography and/or some faults in up-slope structures (e.g., berms and channels used to isolate the hillslopes from outside sources of overland flow such as mining tracks and banks; Moreno-de las Heras *et al.* 2008). This particular feature gives us the opportunity to select two kinds of restored slopes that differ with respect to disturbance level, with 12 slopes that are rilled and 13 that are not.

Soils in natural slopes (unaffected by mining) range from *Typic* or *Lithic Xerorthent* to *Calcic Xerochrept* (*sensu* Soil Survey Staff 1998), and have a low content of organic matter (<3%) and basic pH (Arranz 2004). Most of the natural slopes are covered by sparse shrub communities (dominated by *Genista scorpius* and *Thymus vulgaris*) on abandoned terraces and cereal crops. The natural slopes differ in being either ungrazed or grazed by livestock sheep. Such grazing regime provides us with two types of natural slopes that differ relative to disturbance: three slopes that are grazed seven that are ungrazed.

Measurements of Vegetation and Hydrological Properties Related to Runoff

We measured several metrics of vegetation and soil hydrological properties related to runoff for each of the 35 study hillslopes (25 restored and 10 natural) that were selected to span a broad range of disturbance conditions. For each hillslope, five equidistant 35 m wide transects were located parallel with the slope. Each transect was divided into seven 0.5 m x 0.5 m plots that were separated from each other by 5 m (Fig. 3.1). At each plot, measurements of cover characteristics related surface, vegetation, and overland flow potential were obtained during the spring of 2006. Surface cover was estimated from measurements of fractional cover of bare soil, stone, litter and vegetation. Vegetation cover by species was also estimated visual surveys of the canopy. This sampling procedure for vegetation survey has been successfully tested in reclaimed mining slopes of Mediterranean-dry Spain, encompassing more than 90% of species (Martínez-Ruiz *et al.* 2007). Characteristics of cover

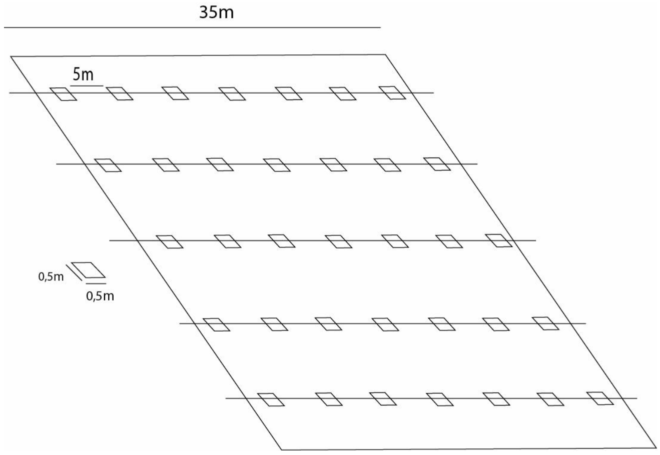


Figure 3.1. Sampling design: hillslopes (n=35, 25 restored and 10 natural; restored slopes were either rilled or not rilled; natural were either grazed or not grazed), each with 5 transects parallel the slope, with each transect having seven 0.5 m x 0.5 m plots that were separated from neighboring plots on the same transect by 5 m.

related to potential for overland flow was assessed using methodology similar to that of Barthès & Roose (2002), and included measurements of four overland flow features: sheet flow, rill flow, ponding areas and infiltration areas. Sheet flow cover was estimated based on cover from surface crusts, stones on the soil surface, small pedestals and microcliffs; rill flow cover was estimated based in cover of grooves, rills and gullies; ponding area cover was estimated based on cover of surface micro-depressions; and infiltration area cover was based on cover of vegetation and litter. We estimated rill erosion rates at the slope scale from the rill network dimensions following the methodology of Morgan (1995). Three composite soil samples (each sample formed by three homogeneously mixed subsamples, randomly distributed in each parallel transect) were taken from the first 15 cm of the soil profile in each slope. Stoniness (%) was determined as the content of soil particles > 2 mm. Accumulated sheet erosion index (ASEI), which accounts for hydrological processes (i.e., the loss of fine soil particles from the soil surface as a consequence of sheet erosion) spanning the duration of the hillslope lifetime, was estimated using the method of Moreno de las Heras *et al.* (2008), calculated as the ratio between mean rock fragment cover of the soil surface (estimated from rock fragment cover measured in $0.25 m^2$ plots) and mean soil stoniness.

Data Analysis

Plot scale analyses

At the scale of individual vegetation patches, which correspond to plot measurements, vegetation cover data were analyzed using a divisive hierarchical classification obtained with Two Way Indicator Species Analysis (TWINSPAN; Hill 1979). The maximum number of indicators per division was five and the classification was followed up to the third division. The "characteristic species" of each group were determined by Indicator Species Analysis (Dufrene & Legendre 1997). In addition, we used nested permutational multivariate analysis of variance (PERMANOVA, Anderson 2001) to identify potential relationships between vegetation communities, as obtained with TWINSPAN, and hydrological properties (overland flow features). The PERMANOVA used the "adonis" procedure in the vegan package (Oksanen *et al.* 2010; R package version 2.9.1; R Development Core Team 2009). The variables used for the test were the four overland flow features covers (sheet flow, rill flow, ponding areas and infiltration areas), and the vegetation communities as the factor nested within slopes. For simplicity, we plotted the hydrological properties in a Principal Component Analysis and we used the first PCA axis to show the trends in overland flow hydrological behaviour for the vegetation communities.

Hillslopes scale analyses

At the larger scale of the hillslopes, each of which encompasses a mosaic of vegetation patches, measurements were classified using two CLUSTER analyses (Tryon 1939): one for vegetation, and one for hydrological properties related to runoff. Variables of both CLUSTER classifications were independent from the variables selected for the characterization of the obtained groups. For each slope, the variables used to identify vegetation groups were vegetation community covers per slope, and the variables used to identify hydrological groups were the four overland flow features covers (sheet flow, rill flow, ponding areas and infiltration areas). Setup parameters were Euclidean distance measure and Ward's group linkage method (Ward 1963).

To evaluate potential association between vegetation and hydrological properties, hydrological groups were evaluated for differences in vegetation properties different from those used in determining the vegetation groups (total vegetation cover, total species richness and Shannon's diversity at slope scale). Similarly, vegetation groups were evaluated for differences in hydrological properties different from those used in determining the hydrological groups (rill erosion rate and sheet erosion index, both measured at slope scale). Significant differences (considered at α=0.05) among groups for a given property were determined by Kruskal-Wallis analysis.

To evaluate the strength of the two-way interrelationships we used a Redundancy Analysis (RDA; R package version 2.9.1; R Development Core Team 2009) which quantifies the percent of variance in vegetation explained by hydrological properties related to runoff. For this, we classified the slopes in four disturbance categories: Restored rilled (12 slopes), restored not rilled (13 slopes), natural grazed (3 slopes) and natural ungrazed (7 slopes) and we performed the RDA for each disturbance group. The variables used in the analysis were species vegetation matrix for vegetation and overland flow features for hydrological properties. Other statistical analyses were performed using STATISTICA (Statsoft 2001).

Results

Plot Scale Analyses of Vegetation, Hydrological Properties Related to Runoff, and Ecohydrological Units

For vegetation at the plot scale, the TWINSPAN analysis identified seven different vegetation communities based on floristic composition of 110 species appearing in more than 5% of the plots (Fig. 3.2a). For its use in subsequent analysis, we designated bare soil as an additional vegetation cover category, resulting in a total of eight vegetation communities. Community composition was significantly related with soil hydrological properties (PERMANOVA, $F_{1,1224}$=38.78, r^2 = 0.18, $P < 0.001$). For hydrological properties related to runoff at the plot scale, the PCA analysis indicated two main gradients (Fig. 3.2b). The first component of the PCA explained 38.81% of the variance and roughly corresponded to gradients of infiltration and sheet flow; the second component of the PCA explained an additional 21.04% and roughly corresponded to rill flow.

We designated "ecohydrological units" based on the eight vegetation community categories and their associated hydrological properties along the main environmental gradient (PCA 1). These ecohydrological units were differentiated into three main hydrological behaviours: sources of runoff, sinks of runoff, and a transition state between these two (Fig. 3.2c). In general, the *sink* group is composed of two vegetation communities (4 and 7) that include mainly phanerophytes and are characterized by high vegetation cover (Fig. 3.2a). One of these sink communities (7) is dominated by *Juniperus oxycedrus* and *Bupleurum fruticescens*, which occurs only on natural hillslopes, and the other (4) is dominated by *Genista scorpius* and

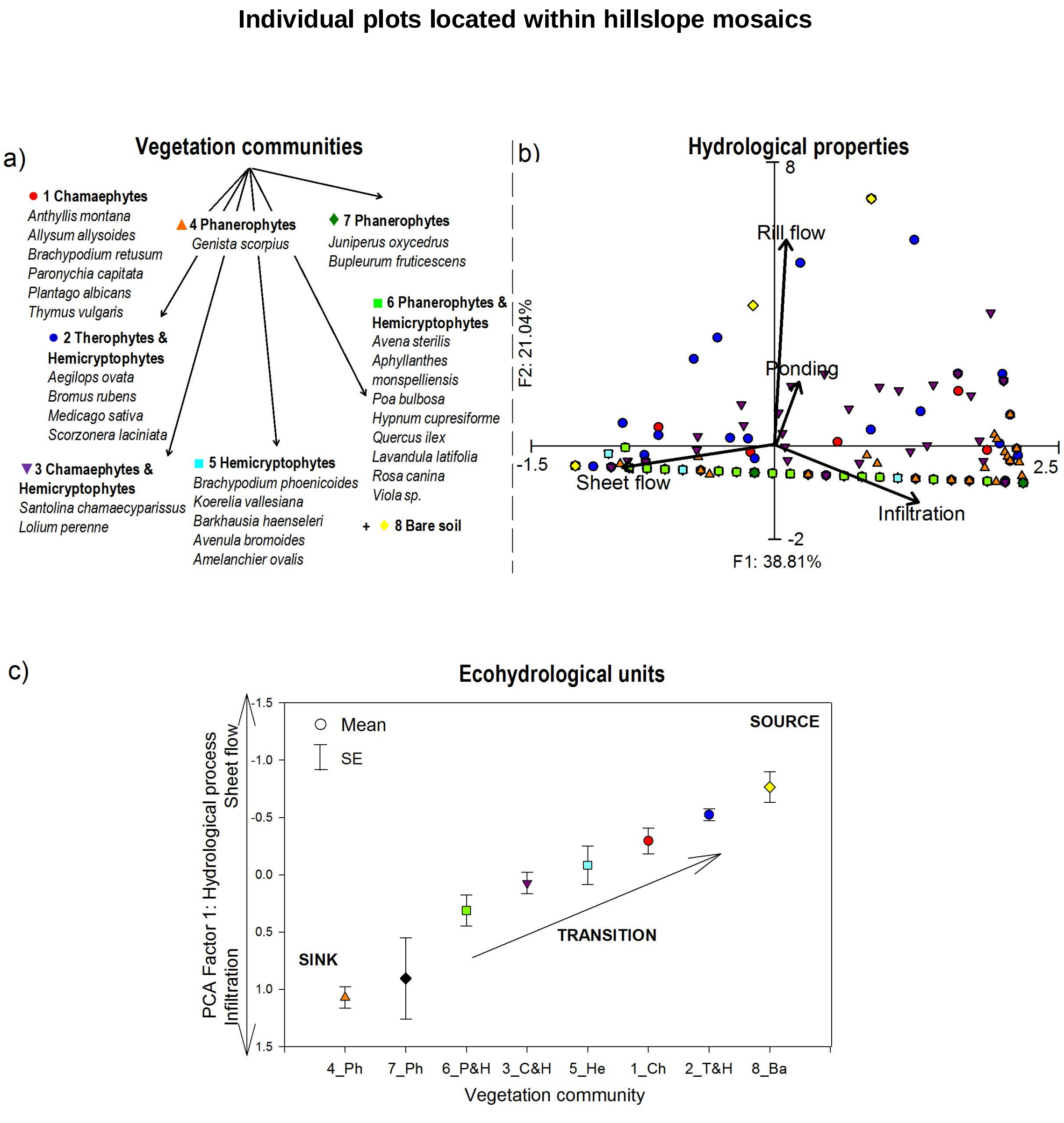


Figure 3.2. Identification of ecohydrological units linking vegetation communities and hydrological properties. a) Vegetation communities identified using TWINSPAN and the characteristic species of each; bare soil was designated as an additional vegetation cover category. b) Hydrological properties identified by PCA ordination, with symbols indicating vegetation community. c) Ecohydrological units identified by PERMANOVA analysis between vegetation communities and hydrological properties (PCA 1): source, sink and transition.

occurs mainly on restored hillslopes. The *source* group is composed of two vegetation communities (2 and 8) that occur primarily on restored hillslopes: one dominated by bare soil (8) and the other characterized by terophytes and hemicryptophytes and having low vegetation covers (2). The third group of ecohydrological units is the *transition* between sources and sinks of runoff, represented by four ecohydrological units (1, 3, 5 and 6) that are characterized by the presence of chamaephytes, hemicryptophytes and phanerophytes and scarcely terophytes, which occur on both natural and restored slopes (see Appendix 3.A).

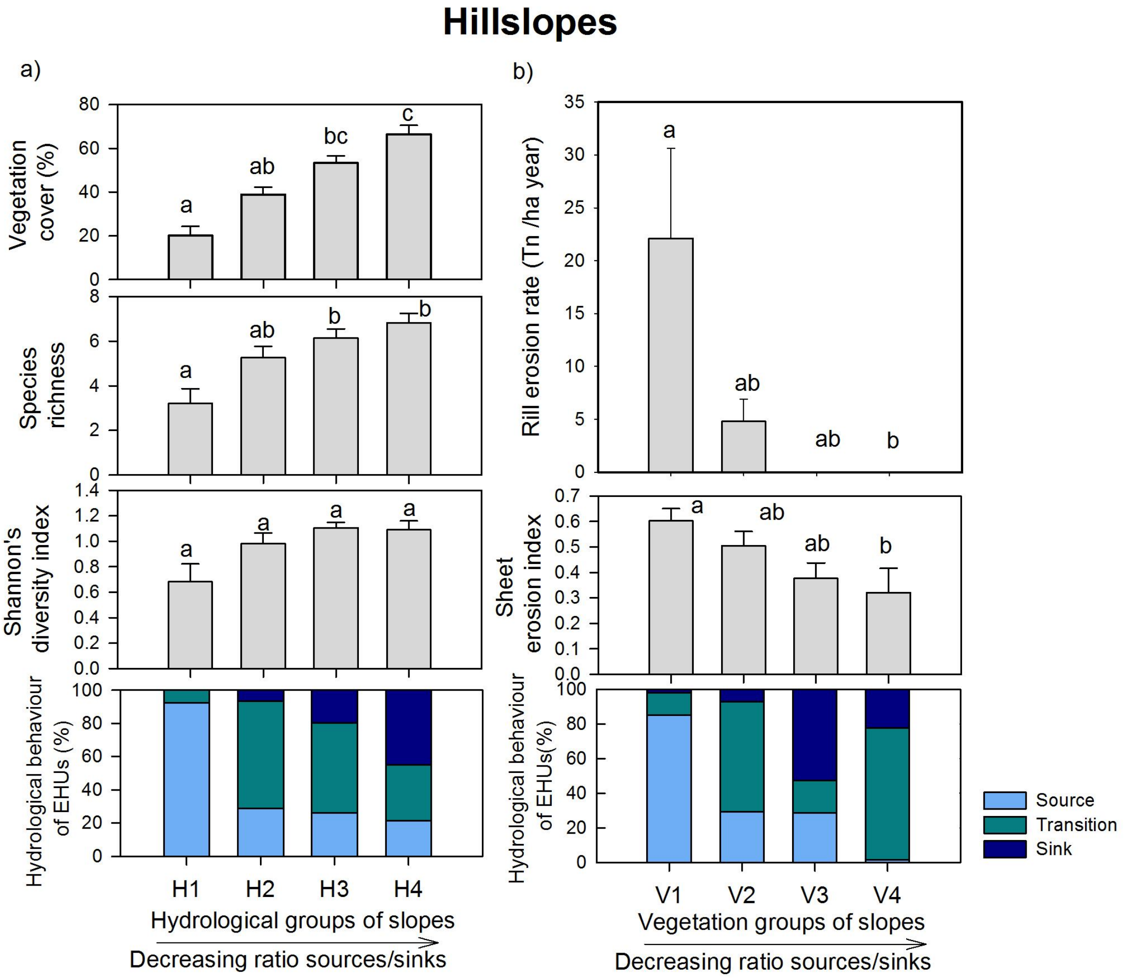


Figure 3.3. Classification of hillslopes based on hydrological properties (a) and vegetation properties (b) using CLUSTER, resulting in 4 groups in either case. Hydrological groups were evaluated for differences in vegetation properties other than those used in determining the vegetation groups (cover, richness and diversity); similarly, vegetation groups were evaluated for differences in hydrological properties other than those used in determining the hydrological groups (rill erosion and sheet erosion). Significant differences among groups for a given property are indicated by different letters (Kruskal-Wallis tests).

Hillslope Scale Analyses of Vegetation and Hydrological Properties Related to Runoff

At the larger scale of the hillslopes, each of which encompasses a mosaic of vegetation patches, measurements were classified using two CLUSTER analyses: one for vegetation and one for hydrological properties related to runoff. Classification of hillslopes using CLUSTER resulted in 4 groups based on either hydrological properties (Fig. 3.3a) or vegetation properties (Fig. 3.3b). Resulting hydrological groups had significant differences among some groups (Kruskal-Wallis) in vegetation properties other than those used in determining the vegetation groups for cover and for species richness, although not for Shannon's diversity (Fig. 3.3a; see Appendix 3.B). These differences are largely reflected on the changes in the proportions of source and sink areas (Fig. 3.3a). Similarly, resulting vegetation groups had significant differences among some groups (Kruskal-Wallis) in hydrological properties other than those used in determining the hydrological groups for both rill erosion and accumulated sheet erosion index (Fig. 3.3b; see Appendix 3.C). These differences are also largely reflected in the changes in the proportions of source and sink areas (Fig. 3.3b).

Ecohydrological Interrelationships Strength Assessed As Vegetation Variation Explained by Hydrological Processes

Ecohydrological interrelationships strength estimated by redundancy analysis (RDA) indicated how much of the variation in vegetation properties was explained by hydrologic properties (Fig. 3.4). Vegetation variability is better explained by hydrology in restored slopes (13.7%) than in natural slopes (3.1%; Fig. 3.4a). Additionally, the variation in vegetation explained by hydrological processes also varies in response to disturbance type (Fig. 3.4b), with the higher amount of vegetation being explained by hydrological processes occurring in restored not rilled slopes (16.7%) and the lowest amount explained for natural ungrazed slopes (3.0%).

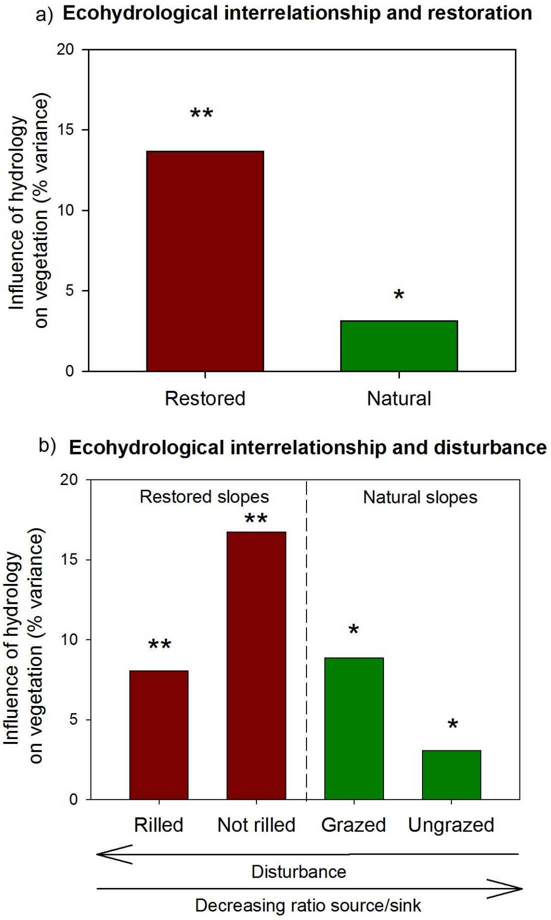


Figure 3.4. Ecohydrological interrelationship strength estimated by Redundancy Analysis (RDA) in each disturbance group to determine how much of the variation in vegetation properties is explained by hydrologic properties along a gradient of disturbance and with respect to the ratio of source to sink area, (a) for hillslopes when aggregated to restored or natural, and (b) for hillslopes that were restored rilled, restored not rilled, natural grazed and natural ungrazed. Statistical significance codes: '**'=0.01; '*'=0.05.

Discussion

Our results at both the patch and hillslope scales, evaluated from both ecological and hydrological perspectives, provide a generally consistent picture of associations between vegetation and hydrological properties leading to runoff, and indicate ecohydrological interrelationships that vary in strength with disturbance. In fact, our results illustrate that species composition at the patch scale corresponds directly with three main types of ecohydrological behaviour: sources of runoff, sinks of runoff and transitions between sources and sinks. Such differentiation between sources and sinks of runoff has been previously shown to influence plant growth and the development of bare patches (Pugnaire *et al.* 1996; Ludwig *et al.* 2005), with runoff from bare source patches being captured as runon in herbaceous patches (Seghieri & Galle 1999; Yu *et al.* 2008; Urgeghe *et al.*, 2010). Other studies have found relationships for runoff and soil erosion with vegetation characteristics of canopy cover (Quinton *et al.* 1997) and plant morphology (Cerdà 1997; Bochet *et al.* 2006). Our results illustrate how species composition, not just amount of vegetation cover, is directly related to hydrological processes. Species composition of individual vegetation patches can therefore be used to infer hydrological properties and *vice versa*. The significant differences in the three types of ecohydrological units identified at the patch scale are apparently substantial enough to drive interrelationships between vegetation patterns and hydrological processes at the hillslope scale. Progressive changes in the proportions of source and sink ecohydrological units are loosely echoed in a variety of vegetation and hydrological characteristics (Fig. 3.3).

The generally consistent results indicating that vegetation characteristics correspond to hydrologic properties and *vice versa* suggest a two-way feedback from hydrology to vegetation and from vegetation to hydrology. The redundancy analyses provide more direct support of such a feedback, consistent with other studies and models of this process (Ludwig *et al.* 2005; McDonald *et al.* 2009). However, our results reveal that the strength of such ecohydrological interrelationships may vary with the degree to which a hillslope is disturbed (Fig. 3.4). Our experimental design, which included hillslopes subject to a range of disturbance regimes, allowed us to evaluate the effect of disturbance in ecohydrological interrelationships. Our results illustrate how the relative strength of ecohydrological interactions in hillslope mosaics decreased with decreasing disturbance, from unrilled restored to grazed natural to ungrazed natural ones (Fig. 3.5). Notably, however, rilled restored hillslopes -the most disturbed type- had weaker ecohydrological interrelationships than unrilled restored ones. Our results from a progression of disturbance regimes (the *'soil erosion disturbance transect'*) highlight a major threshold between rilled (abiotic-control of water) and unrilled slopes. Such behaviour is consistent with previous research describing patterns of overland flow generation and continuity in association with different climatic conditions (the *'climatic transect'*), where the controls over soil erosion exhibited a sharp threshold-type behaviour between abiotic-controlled (arid) and biotic-controlled (humid) systems (Lavee *et al.* 1998). These threshold-like transition in both the soil erosion disturbance and climatic transects is ultimately associated with connectivity features that influence plant water availability in the hillslope. In short, the relative strength of ecohydrological interactions in hillslope mosaics decreased with decreasing disturbance except for rilled hillslopes, likely because parts of the hillslope become isolated from one another (Moreno-de las Heras *et al.* 2010; Espigares *et al.* 2009). Not surprisingly, our results show a low variance explained in all cases, which is a common fact taking into account that the study is not developed under controlled conditions. Our results highlight in general how ecohydrological feedbacks are interrelated with degree of disturbance, and have useful implications for restoration; managers should pay special attention in trying to avoid the rilling process, where vegetation-runoff interrelationships are minimized.

Practical implications

Our results highlight that ecohydrological interrelationships underlie the progression of a restored hillslope toward a less disturbed one,

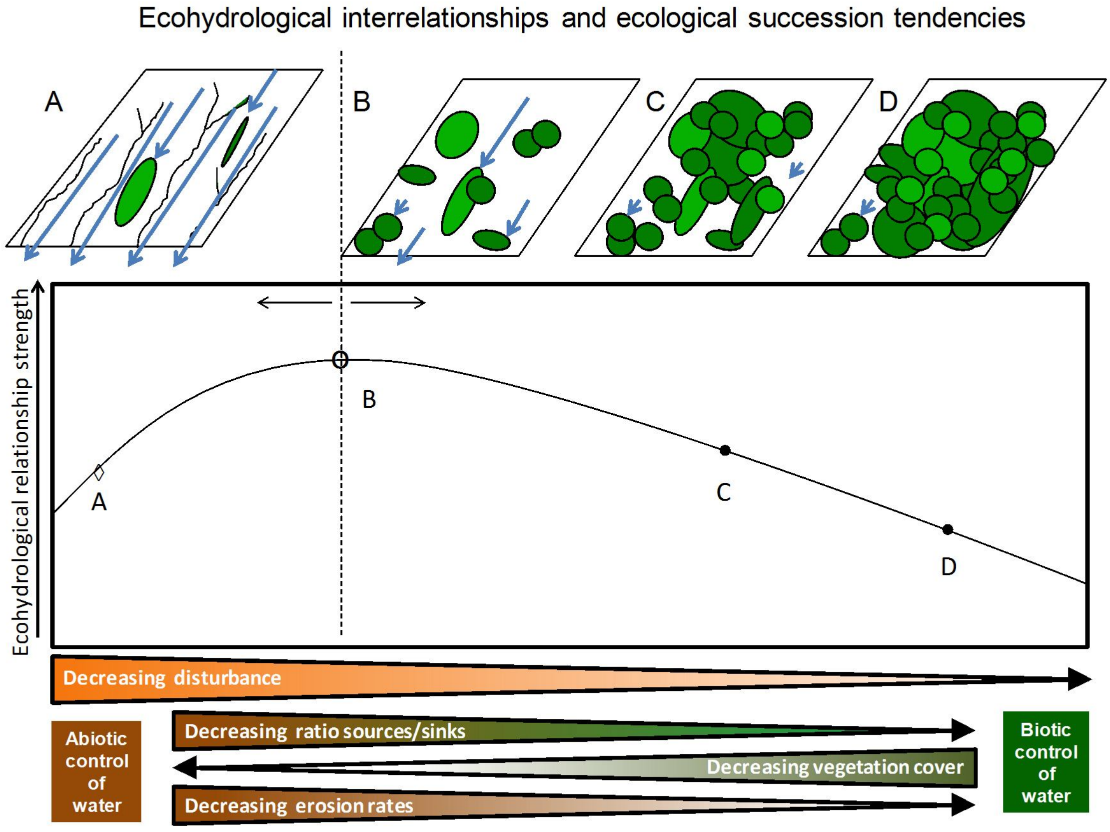


Figure 3.5. Hypothesized trend in which the strength of the ecohydrological interrelationships between vegetation and hydrology varies along a gradient of disturbance spanning from highly disturbed hillslopes having rills (open diamond) to moderately disturbed hillslopes that are not rilled (open circle) to relatively undisturbed hillslopes with higher amounts of vegetation cover (solid circle). The ratio of source to sink area, the amount of vegetation cover, and erosion rates vary with disturbance along the gradient. The strength of the ecohydrological two-way relationship increases from the relatively undisturbed state to more disturbed states until rilling begins, after which the strength of the feedback decreases because parts of the hillslope become isolated from one another. The effectiveness of restoration hinges on preventing rilling with alters the trajectory of future change by the way in which it affects the ecohydrological feedback.

with the interrelationships becoming less strong as more vegetation establishes and the ratio of source areas to sink areas decreases. Indeed, there is an optimum ratio of bare to herbaceous cover that maximizes the total amount of water that herbaceous patches can capture (Yu *et al.* 2008; Urgeghe *et al.* 2010). In fact, when rilling occurs, this trajectory is altered because the connectivity from source patches to sink patches is circumvented by the rills. Consequently, a different type of ecohydrological interrelationship proceeds in rilled hillsopes. Particularly in the case of constructed slopes, land managers should strive to avoid reaching the rilling threshold (open circle at Fig. 3.5). Therefore, caution must be applied when ecohydrological interrelationships are strong because the system can bifurcate toward a weaker feedback either in a desired progression toward a more vegetated and less disturbed state or toward an undesired highly rilled state. Our results provide a key insight into why managing to minimize rill establishment initially is so central to longterm success of restored hillslopes such as mine spoils and other disturbed landscapes.

Acknowledgments

The authors specifically wish to acknowledge J. Fernandez, L. Muñoz for field assistance and J. Rejos for helping with taxonomic determinations. We thank J. Romero-Trillo for assistance with language editing and B. Cuesta and L. Cayuela for their help with the statistics. We are grateful with the Utrillas Council for logistic cooperation. This work was supported by a UAH grant to L. Merino-Martín.

References

Abrahams, A. D., J. P. Anthony, and J. Wainwright. (1995). Effects of Vegetation Change on Interrill Runoff and Erosion, Walnut-Gulch, Southern Arizona. Geomorphology 13:37-48.

Abrahams, A. D., A. J. Parsons, and J. Wainwright. (2003). Disposition of rainwater under creosotebush. Hydrological Processes 17:2555-2566.

Anderson, M. J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology 26:32-46.

Aronson, J., C. Floret, E. Le Floc'h, C. Ovalle, and R. Pontanier. (1993). Restoration and rehabilitation of degraded ecosystems in arid and semiarid lands. I. A view from the south. Restoration Ecology 1:8-17.

Arranz, J. C. (2004) Propiedades, clasificación y evaluación de suelos mineros. Implicaciones sobre la ordenación y gestión de terrenos alterados por la minería. Universidad Politécnica de Madrid.

Barthes, B. and E. Roose. (2002). Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. Catena 47:133-149.

Bautista, S., A. G. Mayor, J. Bourakhouadar, and J. Bellot. (2007). Plant spatial pattern predicts hillslope semiarid runoff and erosion in a Mediterranean landscape. Ecosystems 10:987-998.

Bergkamp, G. (1998). A hierarchical view of the interactions of runoff and infiltration with vegetation and microtopography in semiarid shrublands. Catena 33:201-220.

Bochet, E., J. Poesen, and J. L. Rubio. (2006). Runoff and soil loss under individual plants of a semi-arid Mediterranean shrubland: influence of plant morphology and rainfall intensity. Earth Surface Processes and Landforms 31:536-549.

Bradshaw, A. D. (1983). The Reconstruction of Ecosystems: Presidential Address to the British Ecological Society, December 1982. Journal of Applied Ecology 20:1-17.

Castillo, V. M., M. Martinez-Mena, and J. Albaladejo. (1997). Runoff and soil loss response to vegetation removal in a semiarid environment. Soil Science Society of America Journal 61:1116-1121.

Caylor, K. K., P. D'Odorico, and I. Rodriguez-Iturbe. (2006). On the ecohydrology of structurally heterogeneous semiarid landscapes. Water Resources Research 42:13.

Caylor, K. K., T. M. Scanlon, and I. Rodriguez-Iturbe. (2009). Ecohydrological optimization of pattern and processes in water-limited ecosystems: A trade-off-based hypothesis. Water Resources Research 45:15.

Cerda, A. (1997). The effect of patchy distribution of *Stipa tenacissima* L. on runoff and erosion. Journal of Arid Environments 36:37-51.

Dufrene, M. and P. Legendre. (1997). Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecological Monographs 67:345-366.

Eamus, D., T. Hatton, P. Cook, and C. Colvin, editors. (2006). Ecohydrology : vegetation function, water and resource management. CSIRO Publishing, Collingwood, Vic.

Espigares, T., M. M.-D. L. Heras, and J. M. Nicolau. (2009). Performance of Vegetation in Reclaimed Slopes Affected by Soil Erosion. Restoration Ecology. doi: 10.1111/j.1526-100X.2009.00546.x

Gutierrez-Jurado, H. A., E. R. Vivoni, J. B. J. Harrison, and H. Guan. (2006). Ecohydrology of root zone water fluxes and soil development in complex semiarid rangelands. Hydrological Processes 20:3289-3316.

Hill, M. O. (1979). TWINSPAN--A FORTRAN program for arranging multivariate data in an ordered two-way table by classification of the individuals and attributes. Ecology and Systematics,. Cornell University, NY.

Koch, J. M. and R. J. Hobbs. (2007). Synthesis: is Alcoa successfully restoring a jarrah forest ecosystem after bauxite mining in Western Australia? Restoration Ecology 15:S137-S144.

Lavee, H., A. C. Imeson, and P. Sarah. (1998). The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect. Land Degradation & Development 9:407-422.

Ludwig, J., D. Tongway, D. Freudenberger, J. Noble, and K. Hodgkinson, editors. (1997). Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.

Ludwig, J. A., B. P. Wilcox, D. D. Breshears, D. J. Tongway, and A. C. Imeson. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology 86:288-297.

Macdonald, B. C. T. and M. D. Melville. (1999). The impact of contour furrowing on chenopod patterned ground at Fowlers Gap, western New South Wales. Journal of Arid Environments 41:345-357.

Manu, A., T. L. Thurow, A. S. R. Juo, and I. Zanguina. (2000). Agroecological impacts of five years of a practical program for restoration of a degraded Sahelian watershed. Pages 145-163 in R. Lal, editor. Integrated watershed management in the global ecosystem. CRC Press, Boca Raton.

Martínez-Ruiz, C., B. Fernández-Santos, P. D. Putwain, and M. J. Fernández-Gómez. (2007). Natural and man-induced revegetation on mining wastes: Changes in the floristic composition during early succession. Ecological Engineering 30:286-294.

Mayor, A. G., S. Bautista, E. E. Small, M. Dixon, and J. Bellot. (2008). Measurement of the connectivity of runoff source areas as determined by vegetation pattern and topography: A tool for assessing potential water and soil losses in drylands. Water Resources Research 44:13.

McDonald, A. K., R. J. Kinucan, and L. E. Loomis. (2009). Ecohydrological interactions within banded vegetation in the northeastern Chihuahuan Desert, USA. Ecohydrology 2:66-71.

Michaelides, K., D. Lister, J. Wainwright, and A. J. Parsons. (2009). Vegetation controls on small-scale runoff and erosion dynamics in a degrading dryland environment. Hydrological Processes 23:1617-1630.

Moreno-de las Heras, M., J. M. Nicolau, and T. Espigares. (2008). Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecological Engineering 34:168-178.

Moreno-de las Heras, M., L. Merino-Martín, and J. M. Nicolau. (2009). Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. Catena 77:39-47.

Moreno-de las Heras, M., J. M. Nicolau, L. Merino-Martín, and B. P. Wilcox. (2010). Plot-scale effects on runoff and erosion along a slope degradation gradient. Water Resources Research: 46:W04503, doi:10.1029/2009WR007875.

Morgan, R. P. C. (1995). Soil erosion and conservation. Longman ; J. Wiley, Harlow, Essex, England; New York, NY.

Nicolau, J. M. and M. Moreno-de las Heras. (2005). Open-Cast Mining Reclamation. Pages 370-376 in S. Mansourian, D. Vallauri, and N. Dudley, editors. Forest Restoration in Landscapes: Beyond Planting Trees. Springer, New York.

Oksanen J. , F. Guillaume Blanchet, Roeland Kindt, Pierre Legendre, R. B. O'Hara, Gavin L. Simpson, Peter Solymos, M. Henry H. Stevens and Helene Wagner (2010). vegan: Community Ecology Package. R package version 1.17-2. http://CRAN.R-project.org/package=vegan.

Papadakis, J. (1966). Climates of the world and their agricultural potentialities, Buenos Aires.

Peña, J. L., J. M. Cuadrat, and M. Sanchez. (2002). El clima de la provincia de Teruel. Cartilla turolense. Diputación provincial de Teruel, Teruel.

Puigdefabregas, J. (2005). The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. Earth Surface Processes and Landforms 30:133-147.

Quinton, J. N., G. M. Edwards, and R. P. C. Morgan. (1997). The influence of vegetation species and plant properties on runoff and soil erosion: results from a rainfall simulation study in south east Spain. Soil Use and Management 13:143-148.

R Development Core Team (2009). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Rivas-Martinez, S. (1987). Memoria del mapa de series de vegetacion de España. Ministerio de Agricultura, Pesca y Alimentación, Madrid.

Rodriguez-Iturbe, I. and A. Porporato, editors. (2004). Ecohydrology of Water-Controlled Ecosystems: Soil Moisture and Plant Dynamics. Cambridge University Press, Cambridge, England.

Schlesinger, W. H., J. F. Reynolds, G. L. Cunningham, L. F. Huenneke, W. M. Jarrell, R. A. Virginia, and W. G. Whitford. (1990). Biological feedbacks in global desertification. Science 247:1043-1048.

Seghieri, J. and S. Galle. (1999). Run-on contribution to a Sahelian two-phase mosaic system: Soil water regime and vegetation life cycles. Acta Oecologica-International Journal of Ecology 20:209-217.

Soil Survey Staff. (1998). Keys to Soil Taxonomy. (eds U.-N. R. C. Service). Washington DC.

Statsoft. (2001). STATISTICA. Data Analysis Software System. Tulsa, OK.

Suding, K. N. and R. J. Hobbs. (2009). Threshold models in restoration and conservation: a developing framework. Trends in Ecology & Evolution 24:271-279.

Tongway, D. J., C. Valentin, and J. Seghieri, editors. (2001). Banded vegetation patterning in arid and semiarid environments: ecological processes and consequences for management. Springer-Verlag, New York, USA.

Tryon, R. C. (1939). Cluster analysis. Edwards Brothers, Ann Arbor, Michigan.

Urgeghe, A. M., D. D. Breshears, S. N. Martens, and P. C. Beeson. (2010). Redistribution of runoff among vegetation patch types: On ecohydrological optimality of herbaceous capture of runon. Rangeland Ecology and Management, Submitted.

Ward, J. H., Jr. (1963). Hierarchical Grouping to Optimize an Objective Function. Journal of the American Statistical Association 58:236-244.

Whisenant, S. G., T. L. Thurow, and S. J. Maranz. (1995). Initiatig autogenic restoration on shallow semiarid sites. Restoration Ecology 3:61-67.

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

Wilcox, B. P. and T. L. Thurow. (2006). Emerging issues in rangeland ecohydrology: Vegetation change and the water cycle. Rangeland Ecology & Management 59:220-224.

Yu, M., Q. Gao, H. E. Epstein, and X. Zhang. (2008). An ecohydrological analysis for optimal use of redistributed water among vegetation patches. Ecological Applications 18:1679-1688.

Zea-Cabrera, E., Y. Iwasa, S. Levin, and I. Rodriguez-Iturbe. (2006). Tragedy of the commons in plant water use. Water Resources Research 42:12.

Appendix 3.A. Comparison of hydrological, cover and plant traits associated to the Ecohydrological Units.

	EHU 1		**EHU 2**		**EHU 3**		**EHU 4**	
	Transition chamaephytes		**Source therophytes and hemicryptophytes**		**Transition chamaephytes and hemicryptophytes**		**Sink chamaephytes**	
N	118		434		190		212	
Hydrological features[1]								
Ponding (%)	0.678±3.782	a	1.081±6.375	a	5.516±17.961	a	0.566±3.567	a
Infiltration (%)	23.864±34.029	ab	16.157±29.163	ad	27.789±35.209	b	64.575±39.824	c
Sheet flow (%)	74.102±36.103	ab	80.037±33.132	a	61.789±39.573	b	34.028±40.311	c
Rill flow (%)	0.136±1.473	a	1.949±12.945	a	3.158±17.534	a	0.000±0.000	a
Cover features[1]								
Bare soil cover (%)	23.085±16.006	a	37.326±18.529	b	27.853±15.703	a	10.462±12.614	c
Stone cover (%)	29.593±17.626	a	27.885±16.141	b	21.768±14.900	a	11.189±13.946	ce
Litter cover (%)	4.847±10.915	acde	1.709±4.150	bde	4.116±9.613	abde	5.330±12.705	abde
Vegetation cover (%)	42.525±18.340	a	33.182±23.185	a	46.200±22.400	b	73.075±24.497	cd
Plant traits (%)								
Species richness	6.229±2.416	b	4.855±2.758	c	5.484±2.654	bc	7.434±2.762	a
Shannon´s index	1.106±0.455	cd	0.951±0.533	bc	0.915±0.470	abc	1.074±0.536	cd
Slope appearance								
Naturality index[2]	0.737±0.442	a	0.002±0.048	b	0.026±0.160	b	0.269±0.444	d

	EHU 5		**EHU 6**		**EHU 7**		**EHU 8**	
	Transition hemicryptophytes		**Transition phanerophytes and hemicryptophytes**		**Sink phanerophytes and hemicryptophytes**		**Source bare soil**	
N	81		128		21		41	
Hydrological features[1]								
Ponding (%)	0.099±0.889	a	0.094±1.061	a	0.000±0.000	a	0.000±0.000	a
Infiltration (%)	31.852±44.517	ab	43.469±45.275	b	60.952±47.529	bc	0.000±0.000	d
Sheet flow (%)	68.049±44.454	ab	56.438±45.197	b	39.048±47.529	bc	81.905±8.206	a
Rill flow (%)	0.000±0.000	a	0.000±0.000	a	0.000±0.000	a	18.095±38.206	a
Cover features[1]								
Bare soil cover (%)	15.901±16.742	ac	10.734±17.453	c	11.619±23.695	c	56.048±23.542	d
Stone cover (%)	19.333±21.237	bcd	8.047±13.787	de	4.095±7.416	e	40.095±21.540	f
Litter cover (%)	17.852±24.037	cd	20.859±23.153	c	11.429±20.815	de	3.667±16.060	e
Vegetation cover (%)	46.914±25.675	bcd	60.266±27.593	d	72.857±34.279	d	0.000±0.000	a
Plant traits (%)								
Species richness	5.877±2.320	b	5.594±2.409	bc	3.952±1.962	c	--------±--------	
Shannon´s index	1.249±0.435	d	1.160±0.492	d	0.529±0.493	a	--------±--------	
Slope appearance								
Naturality index[2]	0.630±0.486	a	0.969±0.175	c	1.000±0.000	ac	0.095±0.297	bd

Abbreviations:
N: Number of quadrats with this EHU.
[1] Cover visually estimated in 35 regularly distributed 0,25 m^2 plots per slope during spring 2006.
[2] Index estimated from 1=Natural slope to 0=restored slope/total of restored slopes.
Values with different letters (a-f) within rows differ significantly at α=0,05. Tested using Kruskal-Wallis and Mann-Whitney U tests.

Appendix 3.B. Basic characteristic (mean±SD) of the four slope's groups obtained from CLUSTER analysis of hydrological data.

	H1		H2		H3		H4	
N	7		9		10		9	
Soil traits[1]								
Stoniness[1] (%)	53.056±2.150	a	46.790±3.028	a	47.873±3.784	a	50.931±2.891	a
Coarse sand[1] (%)	5.065±0.597	a	7.787±0.643	a	7.822±0.803	a	6.841±0.470	a
Sand[1] (%)	11.013±0.692	a	16.078±1.357	a	14.813±1.468	a	13.003±1.105	a
Silt[1] (%)	13.479±0.986	a	14.409±1.095	a	14.042±1.205	a	14.332±1.069	a
Clay[1] (%)	17.387±0.987	a	14.936±0.726	a	15.450±0.854	a	14.893±0.970	a
Texture	Clay loam		Clay loam		Clay loam		Clay loam	
EC[1] -w/v:1/2-(dSm^{-1})	848.843±169.443	a	289.038±41.557	b	369.614±76.701	ab	471.280±91.001	ab
pH[1] -H_2O;W/v:1/2-	7.951±0.118	a	8.201±0.083	a	7.806±0.244	a	7.802±0.208	a
Organic matter [1] (%)	0.812±0.295	a	1.318±0.320	ab	1.558±0.252	ab	2.724±0.719	b
Total Nitrogen[1] (%)	0.051±0.010	a	0.107±0.032	ab	0.124±0.022	ab	0.153±0.042	b
Bulk density[2] ($g\ cm^{-3}$)	1.488±0.023	a	1.421±0.043	a	1.358±0.042	a	1.377±0.052	a
Erosion features								
Rill erosion rate[4] ($t\ ha^{-1}yr^{-1}$)	33.613±11.305	a	4.478±1.767	ab	0.113±0.113	b	0.000±0.000	b
Sheet Erosion Index[3]	0.624±0.035	a	0.641±0.066	a	0.362±0.061	ab	0.252±0.049	b
Vegetation traits[5]								
Vegetation cover (%)	20.033±4.272	a	38.857±3.415	ab	53.263±3.329	bc	66.406±4.051	c
Species richness	3.208±0.657	a	5.254±0.513	ab	6.129±0.427	b	6.822±0.426	b
Shannon´s index	0.680±0.142	a	0.981±0.083	a	1.106±0.042	a	1.089±0.071	a
Topographic features								
Slope lenght (m)	76.143±13.943	a	82.356±5.718	a	56.920±10.335	a	67.822±17.138	a
Slope gradient (^{o})	18.857±0.459	a	18.444±0.709	a	18.400±1.176	a	21.556±1.519	a
Water contributing area (m)	4.701±1.559	a	3.194±1.150	ab	0.000±0.000	b	2.871±0.857	ab
Aspect Index[6]	4.857±0.143	a	4.000±0.577	a	4.700±0.300	a	4.889±0.111	a
Naturality index[7]	0.000±0.000	a	0.222±0.147	a	0.500±0.167	a	0.333±0.167	a

Abbreviations:
N: Number of slopes; EC: Electrical conductivity; w/v: relation weigh(soil)/volume (water)

[1] Measured in three composited samples (each formed of three subsamples) in each slope from the first 10 cm, regularly distributed.

[2] Measured in unaltered soil cores (3 cm height by 5 cm diameter) randomly distributed.

[3] Measured by relation: stone cover/stoniness

[4] Measured from rill network dimensions following Morgan (1997).

[5] Cover visually estimated in 35 regularly distributed 0,25 m^2 plots per slope during spring 2006.

[6] Measured in 5 classes from North=5 to South=1

[7] Index estimated from 1=Natural slope to 0=restored slope /total of restored slopes.

Values with different letters (a-f) within rows differ significantly at α=0,05. Tested using Kruskal-Wallis and Mann-Whitney U tests.

Appendix 3.C. Basic characteristic (mean±SD) of the four slope's groups obtained from CLUSTER analysis of vegetation data.

	V1		V2		V3		V4	
N	11		7		7		10	
Soil traits[1]								
Stoniness[1] (%)	50.113±2.951	a	50.503±3.989	a	45.831±2.939	a	50.403±3.058	a
Coarse sand[1] (%)	5.905±0.671	a	7.147±0.550	a	7.062±0.267	a	8.091±0.863	a
Sand[1] (%)	12.597±1.233	a	14.493±1.718	a	15.164±1.377	a	14.078±1.282	a
Silt[1] (%)	14.084±1.027	a	13.159±1.150	a	15.788±1.071	a	13.588±1.032	a
Clay[1] (%)	17.301±0.852	a	14.697±0.847	ab	16.155±0.561	ab	13.839±0.740	b
Texture	Clay loam		Clay loam		Clay loam		Clay loam	
EC[1] -w/v:1/2-(dSm^{-1})	709.647±130.307	a	234.391±28.428	b	586.000±99.812	ab	293.204±34.777	ab
pH[1] -H_2O;W/v:1/2-	7.976±0.088	a	8.328±0.052	c	8.011±0.066	abc	7.564±0.276	ab
Organic matter [1] (%)	0.995±0.234	a	0.891±0.147	a	1.618±0.206	ab	2.914±0.640	b
Total Nitrogen[1] (%)	0.059±0.007	a	0.063±0.007	a	0.086±0.009	ab	0.225±0.033	b
Bulk density[2] (g cm^{-3})	1.490±0.015	a	1.468±0.023	a	1.438±0.011	ab	1.244±0.041	b
Erosion features								
Rill erosion rate[4] (t ha^{-1}yr^{-1})	22.105±8.508	a	4.796±2.092	ab	0.000±0.000	ab	0.000±0.000	b
Sheet Erosion Index[3]	0.603±0.047	a	0.505±0.056	ab	0.376±0.061	ab	0.321±0.096	b
Vegetation traits[5]								
Vegetation cover (%)	30.966±5.792	a	40.800±4.277	ab	61.478±3.053	b	56.366±5.738	ab
Species richness	4.200±0.666	a	5.331±0.518	ab	7.465±0.501	b	5.666±0.275	ab
Shannon´s index	0.813±0.113	a	0.961±0.083	a	1.088±0.094	a	1.115±0.037	a
Topographic features								
Slope lenght (m)	75.773±9.806	a	72.471±5.639	a	66.100±16.525	a	65.030±15.102	a
Slope gradient (°)	18.455±0.474	a	19.143±0.404	a	19.714±0.993	a	20.100±1.853	a
Water contributing area (m)	3.758±1.252	a	2.902±1.165	a	2.546±1.084	a	0.802±0.537	a
Aspect Index[6]	4.818±0.122	a	5.000±0.000	a	4.857±0.143	a	3.900±0.567	a
Naturality index[7]	0.000±0.000	a	0.000±0.000	a	0.000±0.000	a	1.000±0.000	b

Abbreviations:
N: Number of slopes; EC: Electrical conductivity; w/v: relation weigh(soil)/volume (water)

[1] Measured in three composited samples (each formed of three subsamples) in each slope from the first 10 cm, regularly distributed.

[2] Measured in unaltered soil cores (3 cm height by 5 cm diameter) randomly distributed.

[3] Measured by relation: stone cover/stoniness

[4] Measured from rill network dimensions following Morgan (1997).

[5] Cover visually estimated in 35 regularly distributed 0,25 m^2 plots per slope during spring 2006.

[6] Measured in 5 classes from North=5 to South=1

[7] Index estimated from 1=Natural slope to 0=restored slope /total of restored slopes.

Values with different letters (a-f) within rows differ significantly at α=0,05. Tested using Kruskal-Wallis and Mann-Whitney U tests.

Capítulo 4

A Marta y Óscar

Al corazón del amigo
abre la muralla.
Al veneno y al puñal
cierra la muralla.
Al mirto y la yerba buena
abre la muralla.
Al diente de la serpiente
cierra la muralla.
Al ruiseñor en la flor.
abre la muralla.

La Muralla, Ana Belén y Víctor Manuel

Capítulo 4

Heterogeneidad hidrológica de paisajes restaurados de clima mediterráneo-seco: generación de escorrentía y sedimentos de manchas y laderas en un gradiente de escorrentía superficial

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Merino-Martín, L., Moreno-de las Heras, M. Merino-Martín, J.A., Pérez-Domingo S., Espigares, T., Nicolau, J.M. En revisión. Hydrological heterogeneity in mediterranean-dry reclaimed landscapes: runoff and sediment yield of patches and slopes along a gradient of overland flow. Enviado a *Hydrology and Earth System Sciences (HESS)*.

Resumen

La heterogeneidad espacial es un atributo fundamental de los ecosistemas con restricciones hídricas, controlando los flujos de recursos y energía en los paisajes, especialmente en el caso del recurso hídrico. Por ello, los mosaicos fuentes-sumideros son frecuentemente identificados en estos ambientes. Cabe destacar sin embargo la falta de estudios de heterogeneidad espacial hidrológica en ambientes restaurados, donde la sucesión ecológica y la escorrentía interaccionan especialmente. En este trabajo, se realizó un experimento de campo para estudiar el comportamiento hidrológico de manchas y laderas a lo largo de un gradiente de escorrentía en tres laderas restauradas. Los resultados mostraron que en nuestras laderas, que no tenían desarrollada una red de regueros continua, se produjo el patrón de fuentes-sumideros de escorrentía. Estos microambientes hidrológicos estuvieron asociados a siete tipos de coberturas (caracterizadas por especies o comunidades y coberturas). Dos tipos de sumideros fueron identificados: la comunidad del matorral *Genista scorpius* que puede ser considerado como un "sumidero profundo"; mientras que las comunidades de las gramíneas *Brachypodium retusum* y *Lolium perenne* fueron descritas como "sumideros en superficie" o de "derramaderos de escorrentía". Una variedad de fuentes fue también identificada, comprendiendo desde "fuentes extremas" (comunidad de *Medicago sativa*; equivalente a suelo desnudo) a "fuentes reducidas" (áreas de arbustos de *Thymus vulgaris* o matas herbáceas de *Dactylis glomerata* dispuestos de forma dispersa). Finalmente, se identificó la escorrentía circulante por la ladera como un factor controlador de la diversidad hidrológica: según incrementa la escorrentía superficial a escala de ladera, la diversidad hidrológica disminuyó, desarrollando opuestos comportamientos hidrológicos.

Palabras clave: minería, restauración, erosión, manchas de vegetación, patrón en mosaico, fuentes, sumideros, diversidad hidrológica.

Hydrological Heterogeneity in Mediterranean Reclaimed Slopes: Runoff and Sediment Yield at the Patch and Slope scales along a Gradient of Overland Flow

Luis Merino-Martín[1], Mariano Moreno-de las Heras[2], Jose A. Merino-Martín[3],
Silvia Pérez-Domingo[1], Tíscar Espigares[1], Jose M. Nicolau[4]

[1]*Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, Madrid, España.*
[2]*Faculty of Engineering and Built Environment, University of Newcastle, Callaghan, Australia.*
[3]*Escuela Técnica Superior de Ingenieros en Topografía, Geodesia y Cartografía, Universidad Politécnica de Madrid, Madrid, España.*
[4]*Departamento de Agricultura y Economía Agraria, Escuela Politécnica, Universidad de Zaragoza, Huesca, España.*

Abstract

Hydrological heterogeneity is recognized as a fundamental ecosystem attribute in drylands controlling the flux of water and energy through landscapes. Therefore, mosaics of runoff and sediment sinks and source patches are frequently identified in these dry environments. There is a remarkable scarcity of studies about hydrological spatial heterogeneity in restored slopes, where ecological succession and overland flow are interacting. In this study, we conducted a field experiment to study the hydrological behavior of patches and slopes along an overland flow gradient in three reclaimed slopes. We found that runoff generation and routing in the constructed slopes that have not developed continuous rill networks showed a pattern of source and sink areas. Such hydrological microenvironments were associated to seven vegetation patches (characterized by plant species or communities and cover). Two types of sink patches were identified: shrub *Genista scorpius* patches could be considered as a "deep sink", while patches where the graminoids *Brachypodium retusum* and *Lolium perenne* dominate are classified as "surface sinks" or "runoff splays". A variety of sources were also identified spanning from "extreme sources" (*Medicago sativa* patches; equivalent to bare soil) to "poor sources" (areas scattered by dwarf-shrubs of *Thymus vulgaris* or herbaceous tussocks of *Dactylis glomerata*). Finally, we identified the volume of overland flow routing along the slope as a factor controlling hydrological diversity: as overland flow at the slope scale increased, hydrological diversity diminished, developing opposite hydrological roles.

Key words: mining, restoration, erosion, vegetation patches, mosaic, sinks, sources, hydrological diversity.

Introduction

Spatial heterogeneity is a fundamental ecosystem attribute in drylands, controlling the flux of resources and energy through landscapes (Noy-Meir, 1973; Schlesinger *et al*., 1990). Indeed, water-limited landscapes are generally arranged in a mosaic of patches with diverse hydrological behavior, where surface fluxes of water runoff and sediments interact with vegetation dynamics (Lavee *et al*., 1998; Puigdefábregas, 2005). The Trigger-Transfer-Reserve-Pulse (hereafter TTRP) conceptual model developed by Ludwig *et al*. (1997; 2005) represents a useful framework for the understanding of these processes. This approach suggests that these ecosystems are structured as a mosaic of densely vegetated patches, with resource-sink hydrological behavior, interspaced within a bare soil component, with resource-source hydrological behavior. The transfer of water runoff and sediments from the bare source areas to vegetated patches (i.e. sinks) maximizes the use of resources, producing pulses of vegetation growth which increase the ability of vegetation patches to capture surface resource fluxes and hence, enhance water and soil conservation at the landscape level.

Several studies have highlighted the operation of coupled eco-hydrological processes described by the TTRP conceptual framework in many semiarid and arid ecosystems throughout the world, including Niger (Seghieri *et al*., 1997; Bromley *et al*., 1997), Jordan (White, 1969), Australia (Dunkerley and Brown, 1995), USA (Reid *et al*., 1999) and Spain (Calvo-Cases *et al*., 2003). Further work refined the principles of this approach, explaining that the sink-source hydrological behavior of vegetation patches does not only depend on vegetation density, but also on the specific traits of the dominant plant species growing in them (Bochet *et al*., 2006; Garcia-Estringana *et al*., 2010; Vasquez-Mendez *et al*., 2010).

The effects of disturbance on these resource-conserving natural ecosystems are well known: rises in runoff generation and soil erosion, frequently associated to the development of rill or gully networks (Davenport *et al*., 1998; Ludwig and Tongway, 2000; Wilcox *et al.*, 2003). Under these conditions, vegetation patches are not able to capture water and sediments efficiently, causing a significant loss of resources at the landscape scale. These losses reduce vegetation growth and, in consequence, could cause the activation of a long-term self-reinforced degradation process. Similar mechanisms operate in degraded human-made slopes under Mediterranean-dry climate (Nicolau, 2002). The dynamics of these reclaimed ecosystems are modulated by the amount of overland flow routed through the slopes, which in some cases is very high, favored by structurally poor soils, rough topographical designs, and occasionally, the presence of runoff contributing areas (e.g. steep berms, tracks, etc.) located at the top of slopes (Moreno-de las Heras *et al*., 2008; Hancock and Willgoose, 2004). When the amount of overland flow is high, intense soil erosion processes develop, rill erosion being the most significant phenomenon. Rill networks drain runoff away from slopes efficiently, reducing water infiltration, thus increasing the water deficit (Moreno-de las Heras *et al*., 2010). As a result, vegetation dynamics are affected, constraining seed germination, plant establishment and development, and seed production (Espigares *et al*., in press). These mechanisms drive the reclaimed ecosystem towards a highly degraded state, in which a very costly intervention is required to facilitate vegetation recovery (Nicolau, 2003).

To date, the study of the interactions between the dynamics of overland flow and vegetation has focused on degradation processes affecting both natural and reclaimed semiarid environments. Notably lacking are studies that focus on the opposite ecosystem recovery phenomenon. Reclaimed mining environments offer excellent opportunities for the elucidation of the structural and functional dynamics of ecological systems, where soils and vegetation are in general very simple, so that plant communities and hydrological processes represent early stages of ecosystem organization (Aronson *et al*., 1993). In this study, we analyze hydrological processes (i.e. runoff generation and routing and sediment yield) acting at both the patch and slope scales in Mediterranean-dry reclaimed mining slopes with sparse vegetation and with absent or poorly developed (i.e. spatially discontinuous) rill networks. Previous work carried out in the same study area (Moreno-de las Heras *et al*., 2009; Moreno-de las Heras *et al.*, 2010) indicated that under these conditions overland flow is not routed by rill networks, running mostly as sheet flow over the surface, so the vegetation is able to establish successfully on slopes, promoting a wide diversity of vegetation patches dominated by a variety of plant species with different cover. We specifically aimed to (a) evaluate the heterogeneity of hydrological roles developed at the patch and slope scales in water-limited reclaimed slopes under the perspective of the ecosystem recovery processes, and (b) analyze the interaction of this hydrological heterogeneity with the amount of overland flow routed along the slopes.

We state as a fundamental assumption that the different vegetation patches developed in these Mediterranean-dry reclaimed slope systems have a diverse hydrological behavior that can be functionally categorized as sources or sinks of resources, in accordance with the TTRP conceptual framework developed by Ludwig *et al*. (1997; 2005). Furthermore, we hypothesize that the hydrological heterogeneity of these reclaimed ecosystems is modulated by the general amount of overland flow routed along the slopes, increasing the complexity of surface hydrological behavior as the amount of overland flow decreases. In addition, we hypothesize that we can find two mosaic-generating processes along the overland flow gradient: mosaics driven by differential erosion (i. e. Wainwright *et al*., 2002), where plant cover plays a passive role, and mosaics driven by vegetation (i. e. Puigdefábregas, 2005), where vegetated patches become hot-spots of soil and vegetation change.

Study Site and Methods

Study Area

The study site is located within the *Utrillas* coalfield (~1100 m above sea level) in the Iberian Mountain Chain (Spain). This work was carried out in three reclaimed mining slopes located at *El Moral* spoil bank (40º47'50''N, 0º50'26''W) that were selected to span a gradient of overland flow. The climate is Mediterranean-Continental with mean annual temperature of 14 ºC (ranging from a minimum mean daily temperature of 6.7 ºC in December and a maximum mean daily temperature 23.1 ºC in July), with air frost period between October and April. The local moisture regime can be classified as dry Mediterranean (Papadakis, 1966) with mean annual precipitation of 466 mm (mainly concentrated in spring and autumn) and potential evapotranspiration of 759 mm, yielding a hydrological deficit of 292 mm running from

June to October. The mean number of annual rainfall events in the area is ~50, with some convective rainstorms occurring especially in summer, characterized by high rainfall intensities of up to 100 mm in 24 hours (Peña *et al*., 2002).

The slopes were built between 1987 and 1988 by the *Minas y Ferrocarril de Utrillas S.A.* mining company, with between 20º and 30 º inclinations and a layer of 100-250 cm of a clay-loam overburden substratum free of major physicochemical constraints (see Table 4.1 for a detailed description of slopes). Revegetation of slopes was implemented after cross-slope ploughing by sowing a mixture of perennial grasses (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis* and *Lolium perenne*) and leguminous herbs (*Medicago sativa* and *Onobrychis viciifolia*). Although the slopes were restored using the same general procedures, they differed in their subsequent evolution (ie. rilling, vegetation development), due to differences in their geomorphological design, chiefly in the upper part (Moreno-de las Heras *et al.*, 2008; 2009). Indeed, a 40º steep berm integrated at the top of some slopes (Fig. 4.1a and b) acts as a water-contributing area, generating important amounts of overland flow and promoting rill erosion processes. This situation gave us the opportunity to select three reclaimed scenarios subjected to a variable amount of overland flow routed along the slopes since their construction. Although there are no significant differences in soil traits between slopes (Table 4.1), differences in the size of the up-slope water contributing area and its associated rill erosion processes have promoted large differences in vegetation development.

Hydrological measurements

Runoff and sediment yield at slope and patch scales were monitored on these three experimental slopes from October 2007 to December 2008. The hydrological year in the study site lasts from early autumn to the end of the next summer; however, due to the scarcity of rain events during the autumn of 2007, the sampling period was extended until December 2008, including the autumn of 2008.

At the slope scale, naturally delimited catchments (unbounded plots) were selected (Fig. 1.4 c). Therefore, the area of the slope-scale plots differed between the three slopes (498m^2 for slope 511m^2 for slope 2, and 1474m^2 for slope 3). At the foot of each catchment two plastic collectors were installed and a cemented central outlet fed into these collectors. From the outlet, runoff was guided through a pipe into two 200-liter storage tanks connected by a ten-slot runoff divider (Fig. 4.1c).

At the patch scale, 21 gerlach plots were installed, encompassing the different vegetation patches present in slopes. In total, we found seven different types of vegetation patches dominated by different plant species (see Table 4.2 for a detailed description of vegetation cover and soil structural properties of patches): scattered clumps of legumes (*Medicago sativa*) and grasses (*Dactylis glomerata*) in a matrix of bare soil, scattered dwarf shrubs (*Santolina chamaecyparissus* and *Thymus vulgaris*) in a matrix of bare soil, and finally, patches densely covered by grasses (*Lolium perenne* and *Brachypodium retusum*) and shrubs (*Genista scorpius*). Three unbounded Gerlach plots 0.5 m wide (connected to 100 l runoff storage drums) per cover type were established in each slope (Fig. 4.1d).

Catchment areas were visually delimitated on the basis of surface microtopography, and ranged from 1 to 16 m^2. For *Medicago sativa* patches, bounded 3 m long Gerlach plots were used (0.5 m wide; Fig. 4.1e), since the high runoff volumes produced by these patches - located on the steep berm integrated at the top of slopes- compromised the operability of the experimental design. A 3 m length gerlach plot was selected according to previous results

Table 4.1. Descriptive features for the three experimental slopes.

	N	Slope 1		Slope 2		Slope 3	
Date of reclamation		1988		1988		1987	
Topography							
Slope area (m^2)		497.53		510.59		1474.30	
Slope gradient (º)		20		20		20	
Area of *Water-Contributing Area* (m^2)		50.36		22.67		0	
Aspect		North		North		North	
[1]*Soil traits*							
Stoniness (%)	9	39.17 ± 4.54	a	40.50 ± 3.16	a	41.91 ± 3.36	a
Sand (%)	9	44.75 ± 2.64	a	45.56 ± 2.68	a	43.95 ± 2.98	a
Silt (%)	9	28.68 ± 0.27	a	25.19 ± 1.07	a	29.75 ± 1.23	a
Clay (%)	9	26.57 ± 2.38	a	29.25 ± 1.61	a	26.30 ± 1.76	a
Texture	9	Clay loam		Clay loam		Clay loam	
pH -H$_2$O; w/v: ½-	9	8.38 ± 0.25	a	8.32 ± 0.15	a	8.01 ± 0.22	a
EC -w/v: ½- (dS m^{-1})	9	0.31 ± 0.06	a	0.31 ± 0.01	a	0.68 ± 0.42	a
Organic matter (%)	9	1.18 ± 0.25	a	1.55 ± 0.34	a	1.99 ± 0.37	a
CaCO$_3$ (%)	9	7.34 ± 0.44	a	6.17 ± 0.38	a	6.84 ± 0.28	a
[2]Bulk density (g cm^{-3})	27	1.48 ± 0.08	a	1.52 ± 0.03	a	1.43 ± 0.01	a
[3]*Cover features*							
Bare soil cover (%)	105	44.63 ± 3.06	a	32.29 ± 3.94	b	23.71 ± 2.75	b
Stone cover (%)	105	25.54 ± 2.97	a	22.69 ± 1.60	a	21.09 ± 2.57	a
Litter cover (%)	105	5.43 ± 1.81	a	1.09 ± 0.39	b	4.00 ± 1.87	ab
Vegetation cover (%)	105	24.40 ± 2.81	a	43.94 ± 4.07	b	51.20 ± 4.16	b
[3]*Plant traits*							
Species Richness	105	3.83 ± 0.28	a	6.43 ± 0.44	b	9.26 ± 0.55	c
Shannon's index	105	0.80 ± 0.08	a	1.21 ± 0.08	b	1.30 ± 0.08	b
Erosion features							
[4]Sheet Erosion Index	9	0.70 ± 0.09	a	0.59 ± 0.05	a	0.52 ± 0.06	a
[5]Rill density (m m^{-2})	3	0.58		0.00		0.00	
[6]Rill erosion rate[4] (t ha^{-1}yr^{-1})	3	8.41		0.00		0.00	

Abbreviations: N: Number of samples; EC: Electrical conductivity; w/v: relation weight (soil) / volume (water). Values with the same letters (a-c) within rows do not differ significantly at α=0.05. Tested using Kruskal-Wallis and Mann-Whitney *post hoc* tests.

[1] Measured in three composite samples (each formed by three subsamples) from the first 10 cm in three transects regularly distributed along the slope.

[2] Measured in nine randomly distributed unaltered soil cores (3cm height by 5 cm diameter).

[3] Cover, visually estimated in 35 regularly distributed 0,25 m^2 plots per slope during spring 2006.

[4] Measured by the relationship: stone cover/stoniness; following Moreno-del Heras et al. (2008).

[5] Linear rill length (m) measured per surface area (m^2).

[6] Measured from rill network dimensions following Morgan (1997).

obtained in the study site, which showed a low scale-dependency of hydrological responses in plots of this length (Moreno-de las Heras *et al.*, 2010). Runoff collected from plots was measured after each runoff event (runoff-producing events occurring within a 24 h period were considered the same event). Runoff was determined by measuring water level in the container and calculting final volume using the relevant geometric equations. During the study period, no runoff event exceeded the storage capacity of tanks and drums, nor were there any significant losses from tanks attributable to evaporation.The stored runoff was stirred, and a representative 1l sample was taken by filling a 1l plastic bottle from the bottom up in order to obtain an integrated sample. Sediment concentrations were determined by oven-drying (at 105ºC) the collected runoff sample until constant weight was achieved. Precipitation amount and characteristics were measured using an automatic recording rain gauge (GroWeather, Davis®) located about 500 m from experimental slopes. Total precipitation was also recorded using three pluviometers, each located on one experimental slope. According to pluviometer data, spatial variations in precipitation during the study period were negligible.

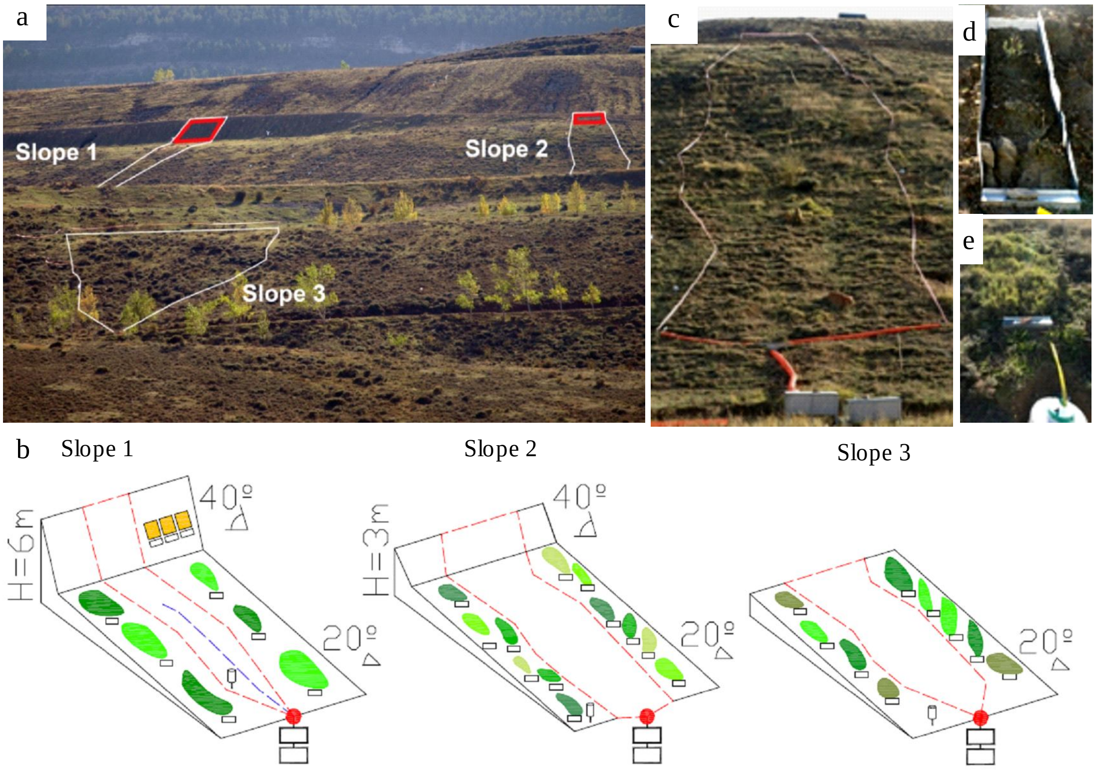
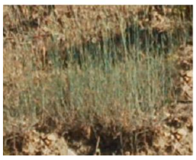


Figure 4.1. (a) Location of the three experimental slopes at the *El Moral* spoil-bank (in red colour water contributing areas); (b) schematic representation of the experimental layout at slopes; (c) example of a slope plot on Slope 2. (d,e) examples of gerlach plots on slopes.

Table 4.2. Characteristics of each patch present in the slopes under study.

Vegetation patch		Cover[1] (%)	Bulk Density[2] (g/cm^3)	AWC[3] (%)
	*Genista*	81.33 a	1.13 a	6.40 a
	*Brachypodium*	93.67 a	1.30 ab	8.52 a
	*Lolium*	67.33 a	1.41 bc	8.07 a
	*Thymus*	23.33 ab	1.55 cd	6.92 a
	*Dactylis*	17.33 ab	1.41 bc	9.01 a
	*Santolina*	19.33 ab	1.42 bc	8.79 a
	*Medicago*	2.67 b	1.61 d	9.09 a

Abbreviations: AWC= Available Water Content. Values with the same letters (a-d) within rows do not differ significantly at $\alpha=0.05$. Analyzed with Kruskal-Wallis and Mann-Whitney U *post hoc* tests.

[1] Cover, visually estimated in six 0.25 m^2 plots per vegetation patch in spring 2008.

[2] Measured in unaltered soil cores (3 cm height by 5 cm diameter) in fifteen samples in each vegetation patch.

[3] Measured, as the difference between volumetric water content at field capacity (Ψ=-0.03MPa) and wilting point (Ψ=1.5MPa) in three samples (each formed by three subsamples) from the top 10 cm.

Topographic and microtopographic measurements

A total station was used to carry out a topographical survey by tacheometry, defining the surface of the vegetation patches. Filling points and break lines were defined in order to determine slope topography. In each slope, the presence of micro-topographic structures was registered in 35 50x50cm plots (seven plots regularly distributed in five transects along the slope).

Soil moisture measurements

To test differences in soil water content between patches, TDR (Time Domain Reflectometry) sensors were installed horizontally at 25 cm into the soil in four replicates of each vegetation patch. A TDR cable tester (Tektronix® 1502C), was used to collect the data, following the methodology proposed by Cassel *et al.* (1994), with an accuracy of 94% in soil moisture determination. Soil water content data were collected within 5 days after each precipitation event.

Data Analysis

Differences in runoff and sediment yield at the slope scale were analysed using Kruskal-Wallis and *post hoc* Mann-Whitney non-parametric tests. Differences in runoff, sediment yield and soil water content at the patch scale were analysed using non parametric tests for repeated measures (Friedman test and *post-hoc* Wilcoxon-Nemenyi- McDonald- Thompson test; Hollander and Wolfe, 1999). In order to analyze the influence of rainfall characteristics on the hydrological role of the different patches, we used linear regressions to relate runoff and sediment yield with rainfall properties (depth and intensity). Runoff, soil moisture and sediment yield data were used to perform a cluster analysis (Tryon, 1939) to separate out groups of patches with homogeneous hydrological role. The linkage rule employed was Ward's method and City-block (Manhattan) as distances measures.

We estimated the "hydrological diversity" of each slope by computing the Shannon diversity index, for which we included the hydrological groups obtained from the cluster analysis as species. This diversity index is dependent of both the richness and abundance (vegetation patch) of each hydrological group.

Statistical analyses were performed using STATISTICA (Statsoft, 2001). *Post hoc* Wilcoxon-Nemenyi-McDonald-Thompson tests were performed with the "coin" and "multcomp" packages of the R program (R_Development_Core_Team, 2009) using the code of "Tal Galili", published on r-statistics.com(*http://www.r-statistics.com/2010/02/post-hoc-analysis-for-friedmans-test-r-code)*.

Results

Rainfall characteristics

A total of 74 rainfall events were registered during the study period, accounting for a total rainfall of 703 mm. During the 2007 hydrological year a total of 550.7 mm were registered, which was above the historical average of 466.2 mm reported by Peña *et al*. (2002) for the study area. Seventeen events (23% of the total number of events) produced runoff. Characteristics of the runoff-producing rainfall and hydrological responses to these events at the slope scale are shown in Table 4.3.

Table 4.3. Rainfall, runoff and sediment yield characteristics of slopes.

Event	Date	Depth (mm)	Duration (min)	$^1I_{30}$ (mm/h)	Rf (mm)			Sy (g m^{-2})		
					SL 1	SL 2	SL 3	SL 1	SL 2	SL 3
1	Mar/9/2008	28.86	90	5.5	0.01	0.00	0.00	1.93	0.00	0.00
2	Apr/13/2008	24.22	45	6	0.01	0.01	0.00	0.00	0.57	0.00
3	May/9/2008	120.11	30	15	17.18	1.01	0.18	187.23	3.06	0.18
4	May/16/2008	18.77	285	12	4.35	0.36	0.04	82.53	2.90	0.26
5	May/17/2008	73.68	105	12	9.72	3.94	1.19	125.41	2.71	0.89
6	May/23/2008	36.34	90	9.5	5.36	0.49	0.05	151.03	20.33	0.29
7	May/31/2008	27.25	210	3.5	1.35	0.08	0.00	73.40	0.64	0.00
8	Jun/9/2008	48.45	555	9	13.82	2.60	1.33	13.82	8.53	1.01
9	Jun/29/2008	45.42	270	24.8	4.96	0.25	0.03	93.14	1.64	0.43
10	Jul/17/2008	32.50	210	4.4	4.15	0.82	0.02	619.81	26.82	0.18
11	Aug/31/2008	16.05	60	4	0.84	0.42	0.01	214.64	6.37	0.24
12	Sep/10/2008	23.01	90	2.5	2.72	0.19	0.01	22.60	1.88	0.01
13	Oct/12/2008	13.93	120	7	1.33	0.09	0.00	22.07	0.41	0.03
14	Oct/18/2008	20.79	30	5.5	2.71	0.17	0.01	12.76	0.38	0.08
15	Oct/24/2008	17.56	180	8	6.36	0.71	0.03	63.71	3.32	0.08
16	Oct/28/2008	28.87	765	4.5	7.47	0.67	0.03	8.67	0.32	0.06
17	Nov/2/2008	48.24	840	7	19.76	2.78	0.22	131.77	1.20	0.17

$^1I_{30}$=30 min maximum rainfall intensity. Rf: Runoff (mm). Sy: Sediment yield (g m^{-2}). SL: slope.

Average duration and depth of runoff-producing events was 233.8 min and 36.7 mm, respectively. Intensity of maximum single event varied from 2.5 mm h^{-1} to 24.8 mm h^{-1}.

Runoff and erosion at the patch scale

Total runoff, runoff coefficients and sediment yield differed significantly between cover types ($p<0.0001$, Friedman test, Fig. 4.2). Multiple comparisons between vegetation patches shown in figure 4.2, suggest that there is a gradient with two extremes: one patch (*Genista*) showing lower runoff and sediment production and higher soil moisture and another patch (*Medicago*) with higher runoff and sediment production rates and lower soil water content.

Hydrological role of vegetation patches vs. rainfall characteristics

Significant linear relationships between runoff and rainfall depth between each patch were obtained, showing a variable effect of the amount of rain on runoff (Fig. 4.3a) depending on patch. The slope of the regression equation can be used as a parameter to measure the effect of the different vegetation patches on hydrological processes (Figure 4.3b). In accordance with runoff production results, the lowest slope of the regression equation corresponded to the vegetation patch with lowest runoff rates and sediment production (*Genista* patches), and the highest slope of the regression to the *Medicago* patches (which had highest runoff and sediment yield rates).

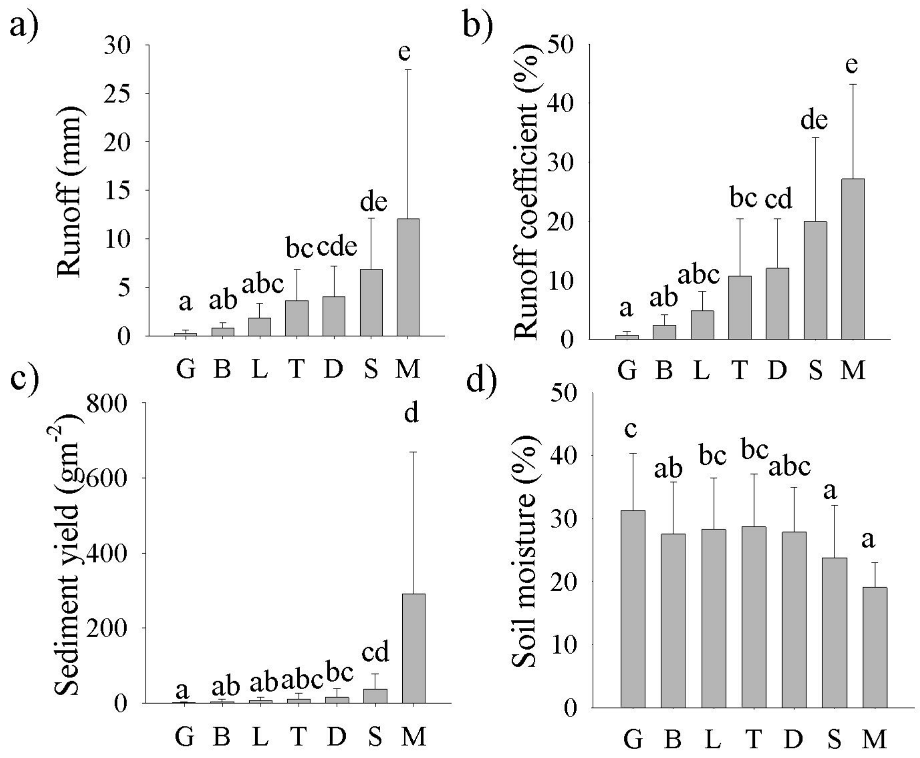
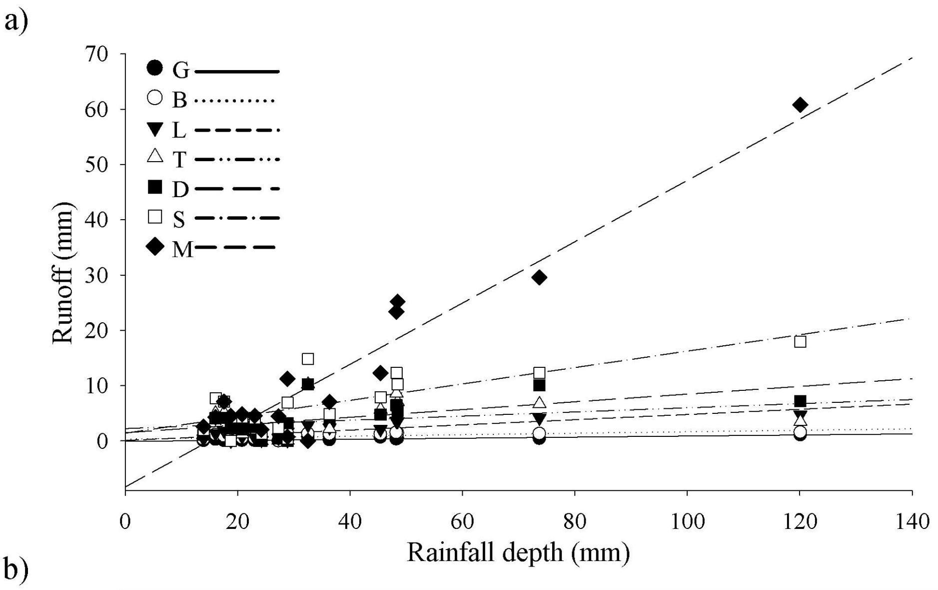


Figure 4. 2. Hydrological behavior of different vegetation patches. G: *Genista*; B: *Brachypodium*; L: *Lolium*; T: *Thymus*; D: *Dactylis*; S: *Santolina*; M: *Medicago*.

Hydrological groups

Classification of vegetation patches based on runoff and soil moisture properties (runoff, runoff coefficient and soil moisture) using cluster analysis resulted in 4 groups (Fig. 4.4). Together with previous results, we related the resulting groups to four main hydrological roles. One group (runoff sinks) included three patches (*Genista*, *Brachypodium*, *Lolium*) with lower runoff and sediment production rates and higher soil water content (Fig. 4.3). In contrast with this group there are two groups with the highest amounts of runoff and sediment yield (*Santolina* and *Medicago*), which can be classified as moderate and extreme runoff sources, respectively. Between these contrasting roles, a fourth group was found (poor runoff response) which included two patches (*Dactylis* and *Thymus*).

Runoff and sediment yield were only recorded for rainfall events that delivered runoff at the foot of slopes (slope-scale runoff producing events). In addition, we used unbounded gerlach plots to describe actual hydrological role of these patches; this approach assumes the fact that some high-intensity precipitation events may have connected a greater surface than the estimated catchment areas for each patch. These two design constraints explain the counterintuitive result of classifying as a runoff sink a surface patch in which some (though very low) amounts of runoff and erosion were recorded.



b)

Vegetation patch	Equation *Runoff* (lm^{-2}) *against Depth (mm)*	p	r^2
Genista	y = -0.0702 + 0.0096*x	0.00006	0.6678
Brachypodium	y = 0.3052 + 0.0134*x	0.00920	0.3734
Lolium	y = 0.0797 + 0.0469*x	0.00007	0.6621
Thymus	y = 2.2381 + 0.0374*x	0.23000	0.0945
Dactylis	y = 1.4735 + 0.0698*x	0.01240	0.3496
Santolina	y = 1.397 + 0.1483*x	0.00060	0.5575
Medicago	y = -8.3062 + 0.554*x	0.00000	0.9043

Figure 4.3. Runoff-rainfall depth relationships obtained for the different patches during the study period (2007-2008). G: *Genista*; B: *Brachypodium*; L: *Lolium;* T: *Thymus*; D: *Dactylis*; S: *Santolina*; M: *Medicago*. b) Linear regression equations relating rainfall depth (mm) with runoff (mm) in the different patches.

Micro-topographic structures

Slope 1, that developed discontinuous rilling processes and higher runoff coefficient and sediment yield rate, presented rill/inter-rill and rill fan micro-environments (Fig. 4.5). In slope 2, runoff splays were found, deposition areas formed by water accumulation in micro-terraces built during restoration practices. In slope 3 no particular micro-topographic structures were found, showing a high abundance of flat forms (Fig. 4.5).

Hydrological heterogeneity at the slope scale

Runoff coefficient and sediment yield differed significantly between the three slopes selected ($p<0.0001$, Kruskal-Wallis test, Fig. 4.6 a, b), showing a gradient of soil erosion from slope 1 to slope 3. Significant linear relationships were found between runoff and rainfall depth at the slope scale, suggesting a different behavior for each of the three slopes (Fig.4.6 c).

When the abundance of each of the four hydrological groups of patches obtained from

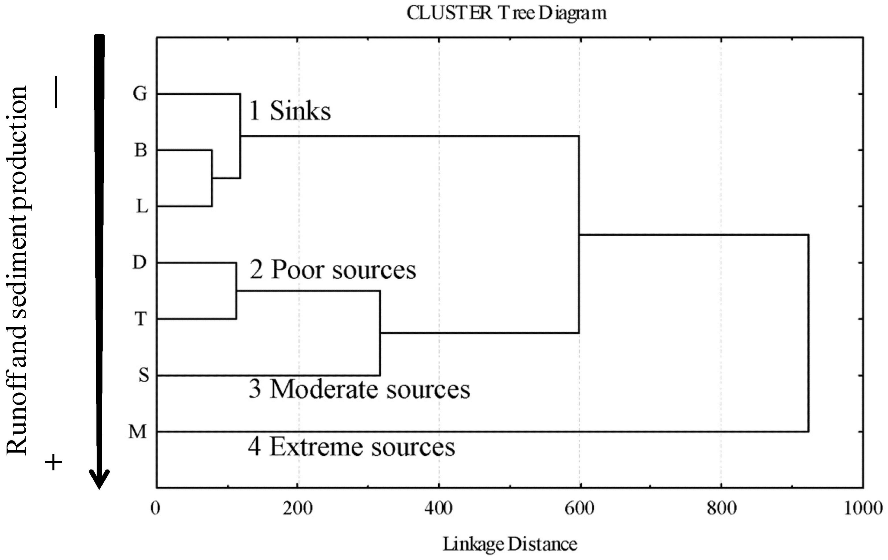


Figure 4.4. Dendogram of the cluster classification analysis with hydrological data of the different patches. G: Genista; B: *Brachypodium*; L: *Lolium*; T: *Thymus*; D: *Dactylis*; S: *Santolina*; M: *Medicago*.

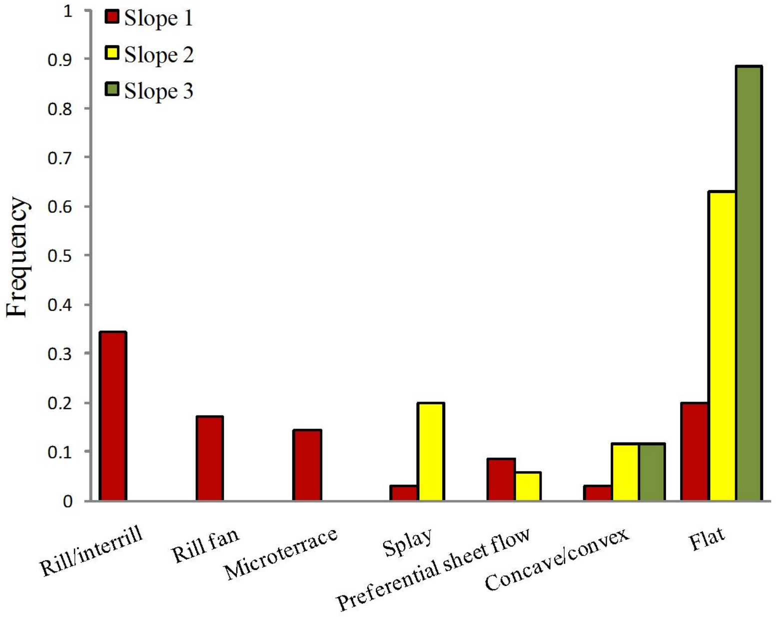


Figure 4.5. Micro-topographic structures found on the three slopes under study.

the cluster analysis in the three slopes were compared (Fig. 4.6 d), extreme sources were found to be absent in slope 3, which was characterized by a fairly homogeneous abundance of moderate and poor sources and sinks. On the contrary, slope 1 was characterized by a dichotomous hydrological role (with extreme and moderate sources and sinks) but without poor source patches that were most abundant in slope 2. Hydrological diversity increased as overland flow decreased (fig. 4.6e).

Discussion

Our results showed that patterns of runoff generation, infiltration and routing in reclaimed slopes are highly dependent on the patch covering the soil, controlling the final role as a runoff source or sink. The cluster analysis applied to field data of runoff and erosion rates and soil water content in the seven patches allowed us to identify four "micro-environments" with different hydrological role: (1) sinks (*Genista*, *Brachypodium*, *Lolium*), (2) poor sources (*Dactylis*, *Thymus*), (3) moderate sources (*Santolina*) and (4) extreme sources (*Medicago*). These findings are supported by comparisons of runoff, sediment yield rates and soil water content, and by the identification of different hydrological responses of patches to rainfall characteristics.

Our results suggest that the general model developed for semi-arid areas (TTRP) describing two main types of hydrological microenvironments (sources and sinks) should be delved into in the case of constructed slopes. Bare patches between plants - characterised by a poor soil structure and a low infiltration capacity- act as runoff generating areas; while areas under plant clumps function as runoff sinks, where organic matter contents are higher, favouring soil aggregation and soil faunal activity, hence increasing macro-porosity and infiltration rates (Calvo-Cases *et al*., 2003; Ludwig and Tongway, 1995; Sánchez and Puigdefábregas, 1994). In our study, we have also identified the type of plant assemblages that play either role, as well as the intensity of that role in each case.

In the studied slopes, sinks were associated with the collection of very low amounts of runoff and sediments dowslope from the patch (Fig. 4.2; G, B, L). Furthermore, these low runoff and sediment yield rates were relatively constant in these patches, even under high precipitation or intense rainfall conditions. These patches are soil surface areas densely covered by grasses (*Lolium perenne, Brachypodium retusum*) or shrubs with a dense herbaceous understory (*Genista scorpius*). *B. retusum* is a common species following disturbance in Mediterranean environments (Bautista *et al*., 1996; Cerdà, 1998). It has been described as a good soil protector, with a high erosion control capacity (De Baets *et al*., 2007) given by a dense root system and the consolidation of soil aggregates (Cerdà, 1998). We found that runoff and erosion rates where low for *Brachypodium* patches, contrasting with the low amounts of soil water content after rainfall at 25 cm, which could suggest that this species is highly efficient in obstructing runoff and sediments although it is not so efficient infiltrating water in depth. In fact, field observations showed *Brachypodium* patches splaying rather than retaining the water and sediment flow. Ryegrass (*Lolium perenne*) is not a characteristic species of Mediterranean degraded environments; it was introduced during revegetation practices. The effects of ryegrass in reducing runoff and erosion rates have been

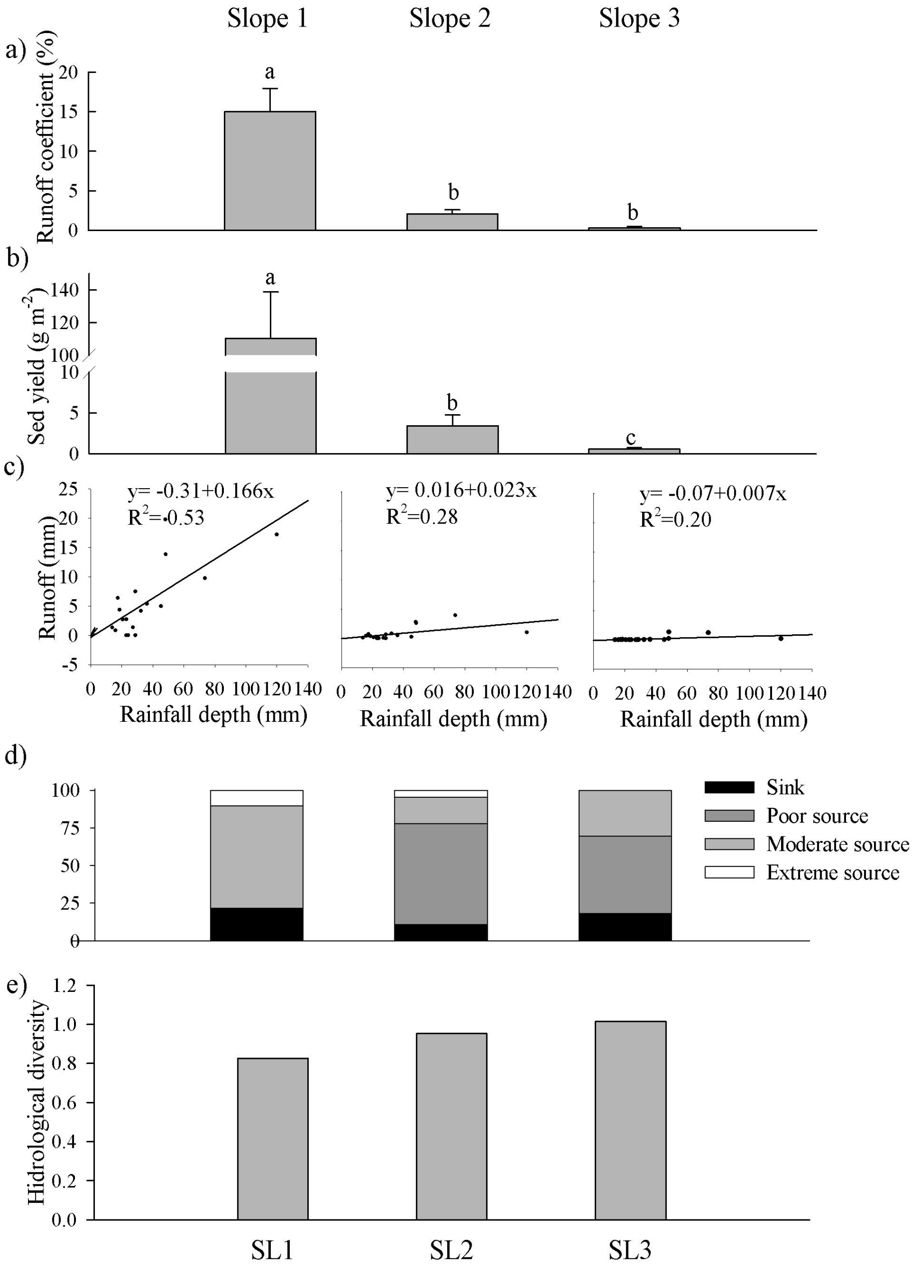


Figure 4.6. Slope hydrology and hydrological heterogeneity. a) Runoff coefficient (%), b) sediment yield (g m^{-2}), c) runoff-rainfall depth relationship, d) hydrological group abundance, e) hydrological group diversity. G: *Genista*; B: *Brachypodium*; L: *Lolium*; T: *Thymus*; D: *Dactylis*; S: *Santolina*; M: *Medicago*.

broadly described (Zhou and Shangguan, 2007). It has been found that canopies of ryegrass usually contribute to runoff declines to a greater extent than roots, whereas roots contributed mainly to a strong decrease in sediment yields (Zhou and Shangguan, 2008). We also found that this patch was highly efficient in obstructing runoff and sediments, although it is the sink with the highest runoff and erosion rates and the most precipitation-dependent hydrological behaviour. *Genista* patches had the significantly lowest runoff and erosion rates and highest soil water content. Soil moisture content after rainfall, which is a good indicator of a sink of runoff behavior, suggests that this patch is not only efficient in obstructing runoff and sediments; it also incorporates water at depth. We found that the soil characteristics of *Genista* understorey are significantly different, with lower bulk density and surface strength. The deeply infiltrated water is easily conserved against evaporation and used by *G. scorpius* and associated plants during dry periods. Molinillo *et al.* (1997) found also that under a dense *G. scorpius* cover, both runoff and sediment yield are strongly controlled. Briefly, our results emphasize the presence of a variety of sink roles dependent on plant community characteristics; *Genista* patch could be considered as a "deep sink", while *Brachypodium* and *Lolium* patches could be described as "surface sinks" or "runoff splays".

Our results suggest an interrelation between micro-topographic structures and vegetation patches present on these microstructures. The three selected slopes represent a gradient of overland flow from discontinuous rilling to very low amounts of runoff routing along the slopes. This gradient generated different micro-topography through soil erosion: rills and rill-fans in the slope with highest runoff volumes (slope 1), splays when runoff volumes decreased (slope 2), and absence of micro-topographic structures in the slope with low runoff volumes (slope 3) (Fig. 5). These results are supported by the fact that two of the three types of sink patches described (*Lolium* and *Brachypodium*) are related to micro-structures: in slope 1 we found *Lolium* patch and in slope 2 we found *Brachypodium* patch (field observations related *Lolium* to rill fans and *Brachypodium* to splays). Thus, we suggest that the proliferation of these cover types was related, in the early stages of succession, to the existence of these micro-structures. In fact, the characteristic species *L. perenne* is a residual species from initial revegetation practices subsisting under favourable conditions generated by rill fans and *B. retusum*, which is a pioneer species typical in degraded environments, would have colonized the splays where it can persist. These findings are in accordance with Wainwright *et al*. (2002), who worked in flows with a discontinuous pattern, with alternating areas of channelization and deposition and found that rill fans (they called these areas "beads") were areas with higher concentrations of nutrient, water and seed resources, finding these sites favourable for the growth of "islands of fertility" (Garner and Steinberger, 1989). However, these findings are not supported by results for slope 3, where different runoff volumes have not eroded the surface differentially and did not generate different micro-structures. Therefore, we suggest that the colonization of the *Genista* sink patch that occurs in this slope is independent of the generation of favourable microenvironments and their initial spatial distribution was random. Thus, we can say that we found the two mosaic-generating processes named by Puigdefábregas (1999): mosaics driven by differential erosion, where plant cover plays a passive role (slopes 1 and 2), and mosaics resulting from "nucleation" processes (slope 3), where vegetated patches become hot-spots of soil and vegetation change. Besides, we found that the gradient of overland flow routing along the slope influences these mosaic generation processes, shifting the driving force of mosaic generation towards an abiotic control under higher overland flow volumes.

In contrast with the sink behaviors described, *Medicago* and *Santolina* patches played an obvious role as sources. *Medicago* patches had the highest runoff rates and sediment yield (considered extreme sources of runoff), followed by *Santolina chamaecyparissus* (moderate runoff source; fig. 4.2). *Medicago* patches have a high bare soil cover (vegetation cover <3%) while *Santolina* patches include vegetated interpatches of scattered dwarf-shrubs (vegetation cover ~20%). Similar runoff and erosion rates to those obtained for Medicago plots were obtained in bare soil by other authors (Cerdà, 1997).

Within these contrasting hydrological roles we found two patches (*Thymus* and *Dactilys*) which belong to vegetated inter-patches areas with cover ranging from 15 to 25%. These areas are covered by scattered dwarf shrubs (*Thymus vulgaris*) or tussocks of orchardgrass (*Dactylis glomerata*). Both species are characteristic of semi-natural Mediterranean environments. *T.vulgaris* develops on relatively erodible soils (Cerdà, 1998), whereas the growth performance of *D. glomerata* has been described as good for erosion control, although not as good as than *L. perenne* (Gokbulak, 2003). However, they play a similar hydrological role which could be explained by differences in cover (23.33% for *Thymus* and 17.33% for *Dactylis*).

Our results confirm that not only plant cover plays a main role for hydrological control in inter-patch areas, but plant morphology is also important. We found higher erosion rates for *Santolina* patches, with similar vegetation cover, than those obtained for *Thymus* or *Dactylis*. This result suggests that there is an interaction between species composition and vegetation cover, as suggested by Bochet *et al*. (2006). They found that rates of soil loss and runoff reduction varied strongly between three different Mediterranean species because of different plant morphology and features. They also found that although canopy cover played a major role for runoff and soil loss reduction, the presence of a second layer of litter cover was fundamental for erosion control during intense rainfall events.

Our experimental design, which included three slopes subjected to the influence of a range of upslope runoff generation, allowed us to evaluate the effect of overland flow on hydrological heterogeneity. The particular influence of runon on ecohydrology of these constructed systems must be taken into account. Moreno de las Heras *et al*. (2009) reported a trend towards hydrological and ecological simplification when run-on cause the development of dense rill networks. Our results, referred to non-concentrated overland flow regimes, illustrate how when the amount of overland flow routing along slopes decreases, hydrological heterogeneity increases. We obtained the highest runoff and erosion rates for slope 1, which developed some discontinuous rills, finding significant differences with the other two non-rilled slopes (Fig. 4.6a, b). Moreover, we obtained very different rainfall-runoff relationships for the set of experimental slopes (Fig. 4.6c), finding that slope 1 is highly dependent on rainfall volume, which together with the previous result, suggests a greater hydrological connectivity of this slope compared with slopes 2 and 3. In fact, these results can be explained as a result of the development of discontinuous rills (Figure 4.1c), which increase runoff connectivity and consequently, runoff and soil erosion rates (Nicolau, 2002; Bracken and Croke, 2007).

The proportion of runoff sources decreased along the overland flow gradient represented by the slopes, the extreme sources being absent in slope 3, that with the lowest runoff coefficient. Moreover, this slope presented the highest values of "hydrological diversity" (an index influenced by both the richness and proportion of each hydrological group). This slope hosts conditions for the development of a very efficient sink, *Genista* patches with a dense understorey of grasses and forbs. This sink shows a remarkable capacity for overland flow interception as well as for increasing soil water content at 25 cm depth. These properties facilitate the development of vegetation and the spread of plant cover, leading the reclaimed ecosystem towards more complex states. Thus, we can say that as upslope generation of overland flow becomes less important in theses slopes, hydrological heterogeneity increases and is driven by vegetation, promoting a variety of sinks and low runoff production areas. Moreover, hydrological connectivity decreases, and water availability for plants increases, reinforcing the trend towards the biological control of the hydrological processes.

Our experimental design also allows us to discuss these results under the perspective of the temporal evolution of sink-source patterns as ecological succession is underway in these reclaimed ecosystems. This phenomenon has been scarcely studied: most previous studies depict hydrological heterogeneity for stable systems throughout the world where sources and sinks are coupled under a dynamic equilibrium state (Ludwig *et al*., 1997; 2005), or address the stability of coupled systems under several types of disturbances, mainly fires and overgrazing, which reduce vegetation cover, and thus, runoff obstruction, increasing runoff and erosion rates (Wilcox *et al*., 2003; McIvor *et al*., 1995; Scanlan *et al*., 1996). Our research describes the variation of the sink-source pattern in a gradient of ecological recovery after slope reclamation. Thus, when upslope runoff generation decreases and the vegetation is able to reach a higher level of complexity, a new type of sink develops: the *Genista* deep sink, with a dense plant understorey. These sink patches significantly increase the biological control of hydrological processes in reclaimed slopes. Therefore, we found that the TTRP framework proposed by Ludwig *et al*. (2005) could be useful for the further understanding of the ecology of these slopes, although more research is needed to investigate whether the water accumulated in the patches provides a pulse for vegetation growth.

Conclusions

Runoff generation and routing in constructed slopes that have not developed continuous rill networks showed a pattern of source and sink areas. Such hydrological microenvironments were associated to seven patches (characterized by plant species or communities and cover). Two types of sink patches were identified: *G. scorpius* clumps, considered as a "deep sink"; *B. retusum* and *L. perenne* described as "surface sinks" or "runoff splays". Surface sinks or runoff splays were related to previous micro-structures while the deep sinks were not related to a previous microtopography. A range of sources were also identified, spanning from "extreme sources" of *M. sativa* (equivalent to bare soil) to "poor sources": areas with scattered dwarf shrubs (*T. vulgaris*) or herbaceous orchardgrass tussocks (*D. glomerata*).

The volume of overland flow routing along the slope controls hydrological diversity. As overland flow at slope scale increases, hydrological diversity decreases and is driven by overland flow, developing contrasted hydrological behaviours (extreme sources and sinks). In fact, when runoff generation in the upper part of the slope stops, the tempered roles (poor sources) become more abundant and a qualitative change occurs and the deep sink formed by patches of *G. scorpius* emerges, hydrological diversity being driven by vegetation in this case. This trend towards an increase in hydrological diversity and the development of deep sinks reflects the evolution through time of the constructed slopes when ecological succession is not "arrested" by overland flow. Thus, in this case, the described pattern of runoff sinks and sources is not stable, but evolves towards a greater biological control of hydrological processes.

Acknowledgements

This work was supported by the Universidad de Alcalá and the project REMEDINAL (S2009AMB-1783), funded by the Regional Government of Madrid. We are grateful to the Utrillas Coucil for their active collaboration. We are also grateful to Oscar Godoy and Sara Godoy for their fieldwork help and Lucia Gálvez for language editing.

References

Aronson J., Floret C., Le Floc'h E., Ovalle C. & Pontanier R. (1993). Restoration and rehabilitation of degraded ecosystems in arid and semiarid lands. I. A view from the south. Restoration Ecology, 1, 8-17.

Bautista S., Bellot J. & Vallejo V.R. (1996). Mulching treatment for postfire soil conservation in a semiarid ecosystem. Arid Soil Research and Rehabilitation, 10, 235-242.

Bochet E., Poesen J. & Rubio J.L. (2006). Runoff and soil loss under individual plants of a semiarid Mediterranean shrubland: influence of plant morphology and rainfall intensity. Earth Surface Processes and Landforms, 31, 536-549.

Bracken L.J. & Croke J. (2007). The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. Hydrological Processes, 21, 1749-1763.

Bromley J., Brouwer J., Barker A.P., Gaze S.R. & Valentine C. (1997). The role of surface water redistribution in an area of patterned vegetation in a semi-arid environment, south-west Niger. Journal of Hydrology, 198, 1-29.

Calvo-Cases A., Boix-Fayos C. & Imeson A.C. (2003). Runoff generation, sediment movement and soil water behaviour on calcareous (limestone) slopes of some Mediterranean environments in southeast Spain. Geomorphology, 50, 269-291.

Cassel D.K., Kachanoski R.G. & Topp G.C. (1994). Practical considerations for using a TDR cable tester. Soil Technology, 7, 113-126.

Cerdà A. (1997). The effect of patchy distribution of Stipa tenacissima L on runoff and erosion. Journal of Arid Environments, 36, 37-51.

Cerdà A. (1998). Soil aggregate stability under different Mediterranean vegetation types. Catena, 32, 73-86.

Davenport D.W., Breshears D.D., Wilcox B.P. & Allen C.D. (1998). Viewpoint: Sustainability of pinon-juniper ecosystems - a unifying perspective of soil erosion thresholds. Journal of Range Management, 51, 231-240.

De Baets S., Poesen J., Knapen A., Barbera G.G. & Navarro J.A. (2007). Root characteristics of representative Mediterranean plant species and their erosion-reducing potential during concentrated runoff. Plant and Soil, 294, 169-183.

Dunkerley D.L. & Brown K.J. (1995). Runoff and Runon Areas in a Patterned Chenopod Shrubland, Arid Western New-South-Wales, Australia - Characteristics and Origin. Journal of Arid Environments, 30, 41-55.

Espigares T., Heras M.M.-d.l. & Nicolau J.M. (in press). Performance of Vegetation in Reclaimed Slopes Affected by Soil Erosion. Restoration Ecology, doi: 10.1111/j.1526-100X.2009.00546.x.

Garcia-Estringana P., Alonso-Blazquez N., Marques M.J., Bienes R. & Alegre J. (2010). Direct and indirect effects of Mediterranean vegetation on runoff and soil loss. European Journal of Soil Science, 61, 174-185.

Garner W. & Steinberger Y. (1989). A proposed mechanism for the formation of fertile islands in the desert ecosystem. Journal of Arid Environments, 16, 257-262.

Gokbulak F. (2003). Comparison of growth performance of Lolium perenne L., Dactylis glomerata L. and Agropyron elongatum (Host.) P. Beauv. for erosion control in Turkey. Journal of Environmental Biology, 24, 45-53.

Hancock G.R. & Willgoose G.R. (2004). An experimental and computer simulation study of erosion on a mine tailings dam wall. Earth Surface Processes and Landforms, 29, 457-475.

Hollander M. & Wolfe D.A. (1999). Nonparametric Statistical Methods. Second edn. John Wiley & Sons, Inc, New York.

Lavee H., Imeson A.C. & Sarah P. (1998). The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect. Land Degradation & Development, 9, 407-422.

Ludwig J. & Tongway D. (2000). Viewing rangelands as landscape systems. In: Rangeland desertification (eds. Arnalds O & Archer S). Kluwer Academic, Dordrecht, The Netherlands pp. 39-52.

Ludwig J., Tongway D., Freudenberger D., Noble J. & Hodgkinson K. (eds.)

(1997). Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.

Ludwig J.A. & Tongway D.J. (1995). Spatial-organization of landscapes and its function in semiarid woodlands, Australia. Landscape Ecology, 10, 51-63.

Ludwig J.A., Wilcox B.P., Breshears D.D., Tongway D.J. & Imeson A.C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology, 86, 288-297.

McIvor J.G., Williams J. & Gardener C.J. (1995). Pasture management influences runoff and soil movement in the semiarid tropics. Australian Journal of Experimental Agriculture, 35, 55-65.

Molinillo M., Lasanta T. & Garcia-Ruiz J.M. (1997). Managing mountainous degraded landscapes after farmland abandonment in the central Spanish pyrenees. Environmental Management, 21, 587-598.

Moreno-de las Heras M., Merino-Martín L. & Nicolau J.M. (2009). Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. Catena, 77, 39-47.

Moreno-de las Heras M., Nicolau J.M. & Espigares T. (2008). Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecological Engineering, 34, 168-178.

Moreno-de las Heras M., Nicolau J.M., Merino-Martin L. & Wilcox B.P. (2010). Plot-scale effects on runoff and erosion along a slope degradation gradient. Water Resources Research, 46, 12.

Nicolau J.M. (2002). Runoff generation and routing on artificial slopes in a Mediterranean-continental environment: the Teruel coalfield, Spain. Hydrological Processes, 16, 631-647.

Nicolau J.M. (2003). Trends in relief design and construction in opencast mining reclamation. Land Degradation & Development, 14, 215-226.

Noy-Meir I. (1973). Desert Ecosystems: Environment and Producers. Annual Review of Ecology and Systematics, 4, 25-51.

Papadakis J. (1966). Climates of the world and their agricultural potentialities, Buenos Aires.

Peña J.L., Cuadrat J.M. & Sanchez M. (eds.)

(2002). El clima de la provincia de Teruel. Instituto de Estudios Turolenses (CSIC), Teruel.

Puigdefábregas J. (2005). The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. Earth Surface Processes and Landforms, 30, 133-147.

Puigdefábregas J., Sole A., Gutierrez L., del Barrio G. & Boer M. (1999). Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in Southeast Spain. Earth-Science Reviews, 48, 39-70.

R_Development_Core_Team (2009). R: A language and environment for statistical computing. In. R Foundation for Statistical Computing Vienna, Austria.

Reid K.D., Wilcox B.P., Breshears D.D. & MacDonald L. (1999). Runoff and Erosion in a Pinon-Juniper Woodland: Influence of Vegetation Patches. Soil Sci Soc Am J, 63, 1869-1879.

Sánchez G. & Puigdefabregas J. (1994). Interactions of Plant-Growth and Sediment Movement on Slopes in a Semiarid Environment. Geomorphology, 9, 243-260.

Scanlan J., Pressland A. & Myles D. (1996). Run-Off and Soil Movement on Mid-Slopes in North-East Queensland [Australia] Grazed Woodlands. The Rangeland Journal, 18, 33-46.

Schlesinger W.H., Reynolds J.F., Cunningham G.L., Huenneke L.F., Jarrell W.M., Virginia R.A. & Whitford W.G. (1990). Biological Feedbacks in Global Desertification. Science, 247, 1043-1048.

Seghieri J., Galle S., Rajot J.L. & Ehrmann M. (1997). Relationships between soil moisture and growth of herbaceous plants in a natural vegetation mosaic in Niger. Journal of Arid Environments, 36, 87-102.

Statsoft (2001). STATISTICA. Data Analysis Software System. In: Tulsa, OK.

Tryon R.C. (1939). Cluster analysis. Edwards Brothers, Ann Arbor, Michigan.

Vasquez-Mendez R., Ventura-Ramos E., Oleschko K., Hernandez-Sandoval L., Parrot J.F. & Nearing M.A. (2010). Soil erosion and runoff in different vegetation patches from semiarid Central Mexico. Catena, 80, 162-169.

Wainwright J., Parsons A.J., Schlesinger W.H. & Abrahams A.D. (2002). Hydrology-vegetation interactions in areas of discontinuous flow on a semi-arid bajada, Southern New Mexico. Journal of Arid Environments, 51, 319-338.

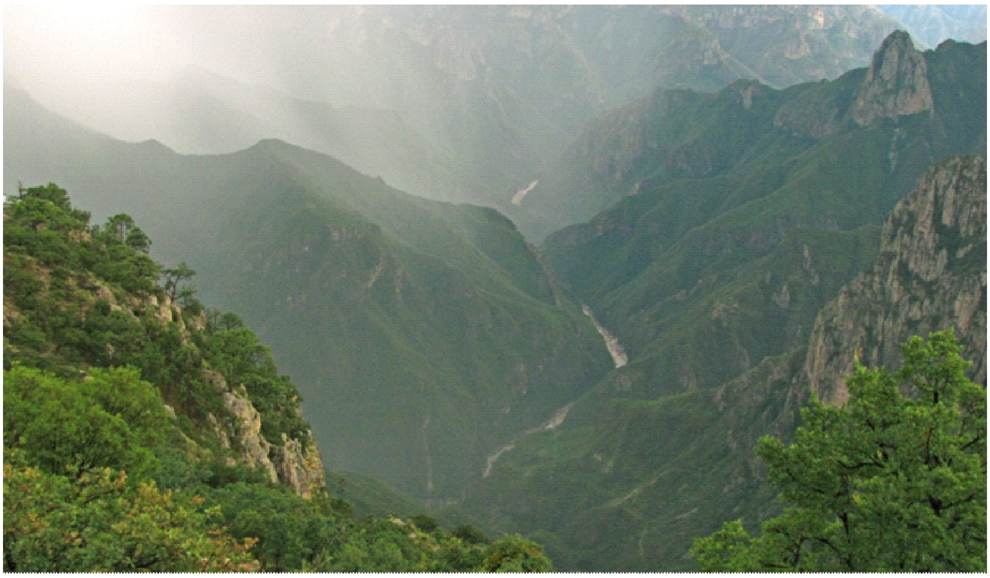
White L.P. (1969). Vegetation Arcs in Jordan. Journal of Ecology, 57, 461-464.

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

Zhou Z.C. & Shangguan Z.P. (2007). The effects of ryegrass roots and shoots on loess erosion under simulated rainfall. Catena, 70, 350-355.

Zhou Z.C. & Shangguan Z.P. (2008). Effect of ryegrasses on soil runoff and sediment control. Pedosphere, 18, 131-136.

Capítulo 5



A Kike, mai fren

Y si seguimos con el plan establecido
nos cansaremos al ratito de empezar
probablemente no encontremos el camino
pero nos sobrarán las ganas de volar.
Improvisemos un guión definitivo
que no tengamos más remedio que olvidar
y acerque todas las estrellas al camino
para que nunca falten ganas de soñar.

El Secreto de las Tortugas, Maldita Nerea

Capítulo 5

La escorrentía como factor director de la sucesión ecológica: evidencias de laderas restauradas en ambientes semiáridos

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Luis Merino-Martín; Mariano Moreno-de las Heras, Silvia Pérez-Domingo; Tíscar Espigares and José Manuel Nicolau. En revisión. Overland flow as a driving force for ecological succession: evidence from reclaimed slopes in semiarid environments. Enviado a *Ecohydrology*.

Resumen

Los ambientes áridos y semiáridos se estructuran frecuentemente en diferentes manchas de vegetación que distribuyen el recurso hídrico de forma heterogénea. La existencia de heterogeneidad hidrológica ha sido estudiada en laderas reguerizadas de ambientes mediterráneos procedentes de la minería de carbón a cielo abierto, donde la restauración es especialmente compleja debido a que es un caso especial de sucesión primaria, junto con el impacto de la escorrentía superficial y con las dificultades para la colonización de la vegetación. En este trabajo, se exploran las interacciones ecohidrológicas a escalas de mancha y de ladera, a lo largo de un gradiente de escorrentía circulante por la ladera. Se siguió la aproximación del marco conceptual del TTRP (Trigger-Transfer-Reserve-Pulse): 1) se compararon los contenidos en humedad del suelo en siete manchas de vegetación; 2) se detallaron diferencias en algunos atributos ecológicos entre manchas; 3) se caracterizaron las interacciones ecohidrológicas con el objetivo de identificar interacciones de retroalimentación; 4) se investigaron los efectos de la escorrentía en las interrelaciones ecohidrológicas. Las hipótesis de partida fueron: 1) el contenido de humedad edáfica está asociado a las manchas de vegetación; 2) un mayor contenido en humedad se relaciona con un mejor desarrollo de la vegetación y de las propiedades del suelo, mejorando el rendimiento de la vegetación y las probabilidades de colonización; y 3) existe una relación inversa entre el volumen de escorrentía que circula por la ladera y el contenido de humedad a escala de ladera, influenciando también procesos a escala de mancha. Los resultados destacan la importancia de la escorrentía circulante por laderas restauradas en la distribución de la humedad edáfica y como consecuencia, en la evolución de las interrelaciones ecohidrológicas para una posterior colonización. En conclusión, puesto que la escorrentía ha sido descrita como un factor director de la sucesión de estos ambientes nuevos debería ser incluida en el diseño de proyectos como un factor clave.

Palabras clave: ecohidrología, minería, restauración, perturbación, humedad edáfica, reguerización discontinua, micro-geomorfología, germinación.

Overland flow as a driving force for ecological succession: evidence from reclaimed slopes in semiarid environments

Luis Merino-Martín[1], Mariano Moreno-de las Heras[2], Silvia Pérez-Domingo[1],
Tíscar Espigares[1] and José-Manuel Nicolau[3]

[1]*Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, Madrid, España.*
[2]*Faculty of Engineering and Built Environment, University of Newcastle, Callaghan, Australia.*
[3]*Departamento de Agricultura y Economía Agraria, Escuela Politécnica, Universidad de Zaragoza, Huesca, España.*

Abstract

Semiarid and arid environments are frequently structured in different vegetation patches that heterogeneously distribute water resources. The existence of hydrological heterogeneity has been studied in Mediterranean restored rilled slopes from opencast coal mining, where reclamation is especially complex because they are a special case of primary ecological succession, together with the impact of overland flow and difficulties for plant colonization. In this study, we explored ecohydrological interactions, both at the patch and the slope scales, along a gradient of overland flow routing along the slope. We followed the TTRP (Trigger-Transfer-Reserve-Pulse) approach: 1) we compared soil moisture content in seven vegetation patches; 2) we described differences in specific ecological attributes between them; 3) we characterized ecohydrological interactions in order to find out feedback interactions; 4) we investigated the effects of overland flow on ecohydrological interrelationships. We hypothesized that: 1) soil moisture content is associated to the vegetation patch; 2) a higher soil water content is associated with a better development of vegetation and soil properties, improving vegetation performance and colonization opportunities; 3) there is an inverse relationship between the volume of overland flow routing and soil moisture content at the slope scale, influencing, in turn, hydrological processes at the patch scale. Results point out the importance of runoff routing along reclaimed slopes by distributing soil moisture and influencing the evolution of ecohydrological interrelationships and subsequent colonization by vegetation. Thus, overland flow should be included in the design of restoration projects as a key factor, as it has been described a driving force for ecological succession in these new environments.

Keywords: ecohydrology, mining, restoration, disturbance, soil moisture, discontinuous rilling, micro-geomorfology, germination.

Introduction

In semiarid environments, where water is the main limiting factor, a strong interrelation between vegetation and hydrology has been widely described (Tongway *et al*. 2001; Puigdefábregas 2005). Frequently, these ecosystems are structured in different vegetation patches that distribute water resources heterogeneously (Cerdà 1997a; Cammeraat & Imeson 1999; Ludwig *et al*. 2000). Particularly, Mediterranean slopes behave as a patchwork of runoff-generating (sources) and run-on-capturing (sink) areas, where the size of the runoff or run-on patches depends on climatic conditions (Lavee *et al*. 1998; Calvo-Cases *et al*. 2003).

The heterogeneity of runoff generation and routing processes in restored slopes from opencast coal mining has also been described in Mediterranean-continental environments (Nicolau 2002). Runoff and, particularly, external run-on coming from the upper part of slopes, have been identified as a significant driving force for vegetation dynamics in these restored slopes (Hancock & Willgoose 2004; Moreno-de las Heras *et al*. 2008). When the magnitude of run-on exceeds a threshold, vegetation recovery is dramatically constrained (Espigares *et al*. in press) and accelerated soil erosion processes arise, being rill erosion the most characteristic phenomenon (Moreno-de las Heras *et al*. 2010). On the one hand, rills efficiently drain runoff away from slopes, reducing rainfall infiltration, and/thus increasing water deficit (Moreno-de las Heras *et al*. 2010). On the other hand, soil moisture content is spatially redistributed, being higher near rills, and lower on inter-rills. As a consequence, vegetation dynamics become severely affected: Espigares *et al*. (in press) have shown how seedling emergence, plant establishment and seed production are limited along a gradient of rilling. Moreno de las Heras *et al*. (2009) showed the link between the spatial pattern of the dominant species (*Medicago sativa*), and soil moisture distribution in rills and interrills.

When the amount of runoff routing downslope is low, overland flow is not directed by rills, running mostly as sheet flow, and the vegetation is able to establish on slopes forming a patchy mosaic structure. In these systems, Merino *et al*. (Chapter 3) described the development of ecohydrological units (classified as runoff sources or sinks) where vegetation and hydrology are strongly associated. Merino *et al*. (Chapter 4) analyzed the hydrological behaviour of different vegetation patches in restored slopes and observed a wide diversity of hydrological responses, from patches that clearly act as runoff sinks ("deep sinks") to runoff source patches characterized by different runoff rates. At the slope scale, a significant relationship between hydrological response and total volume of overland flow routing downslope was also found.

Feedback interactions between vegetation and hydrology in semiarid areas are explained by the Trigger Transfer Reserve Pulse conceptual model (hereafter TTRP) proposed by Ludwig *et al*. (1997). This conceptual framework states that densely co vered vegetation patches obstruct runoff fluxes and store run-on, which promotes plant growth pulses. In turn, vegetation patches enhance soil infiltrability, which is considered a feedback mechanism. This model has been applied for both banded and patchy landscapes (Ludwig *et al*. 2005).

In this study, we explored the interactions between vegetation and hydrological processes both at the patch and the slope scale, along a gradient of overland flow. We followed the TTRP approach, so we started by comparing soil moisture content in vegetation patches; then we analyzed differences in specific ecological attributes between them; and finally we characterized the effects of vegetation patches on soil infiltrability and other soil properties in order to reveal potential feedback mechanisms. Our hypotheses are as follows: 1) Soil moisture content is associated to the type of vegetation patch (source or sink); 2) higher soil water contents are associated with better development of vegetation and soil properties. Specifically, we expected that higher soil water content in sink patches will enhance germination opportunities for plant species; 3) there is an inverse relationship between the amount of overland flow routing and soil moisture content at the slope scale that affects plant germination and colonization; thus the amount of overland flow routing along the slope influences hydrological processes at the patch scale.

Materials and methods

Study area

This study was carried out in three reclaimed mining slopes located at *El Moral* spoil bank (40º47'50''N, 0º50'26''W, Fig. 5.1a). This spoil bank is located in the *Utrillas* coalfield (~1100 m above sea level) in the Iberian Mountain Chain (Spain). The climate is Mediterranean-Continental with a mean annual temperature of 14 ºC (ranging from a minimum mean daily temperature of 6.7 ºC in December and a maximum mean daily temperature of 23.1 ºC in July), with air frost period between October and April. Mean annual precipitation is 466 mm (mainly concentrated in spring and autumn) and potential evapotranspiration is 759 mm, yielding a hydrological deficit of 292 mm running from June to October. The local moisture regime can be classified as dry Mediterranean (Papadakis 1966).

The three slopes were built between 1987 and 1988 by *Minas y Ferrocarril de Utrillas S.A.* mining company. They have a slope gradient of 20º and were covered with a layer of 100-250 cm of overburden clay-loam substratum. Revegetation of slopes was implemented after cross-slope ploughing by sowing with a mixture of perennial grasses (*Festuca rubra, Festuca arundinacea, Poa pratensis* and *Lolium perenne*) and leguminous herbs (*Medicago sativa* and *Onobrychis viciifolia*). Although the slopes were restored using the same general procedures, they diverged in their subsequent evolution (ie. rilling, vegetation development). The triggering factor for these divergent trends is the difference in their geomorphological design, mainly in the upslope section (Moreno-de las Heras *et al*. 2008; 2009) where a steep berm (40º slope) integrated at the top of the slopes works as runoff-contributing area, generating important amounts of overland flow and promoting rill erosion processes (Fig. 5.1a). This special circumstance gives us the opportunity to select three reclaimed slopes subjected to a variable amount of overland flow routed along the slopes. A detailed description of soil, cover, plant and erosion features of the three slopes is presented in table 5.1.

At the patch scale, the different vegetation patches present in slopes were identified first (Fig. 5.1b), characterized by their distinctive species composition and vegetation cover (Merino-Martín *et al*. Chapter 4).

In total, there were seven different patches dominated by different plant species: scattered clumps of legumes (*Medicago sativa*) and grasses (*Dactylis glomerata*) in a matrix of bare soil, scattered dwarf shrubs (*Santolina chamaecyparissus* and *Thymus vulgaris*) in a matrix of bare soil, and finally, patches densely covered by perennial grasses (*Lolium perenne* and *Brachypodium retusum*) and shrubs (*Genista scorpius*). Since *Santolina* patches were the only ones present at both ends of the overland gradient (slopes 1 and 3; Fig. 5.1b) soil moisture and seed bank traits in these patches were sampled twice (once in slope 1 and once in slope 3) in order to explore the consequences of overland flow volume routing along the slopes. The remaining variables were only recorded once at *Santolina*, in slope 1, where the relative abundance of this vegetation patch was higher.

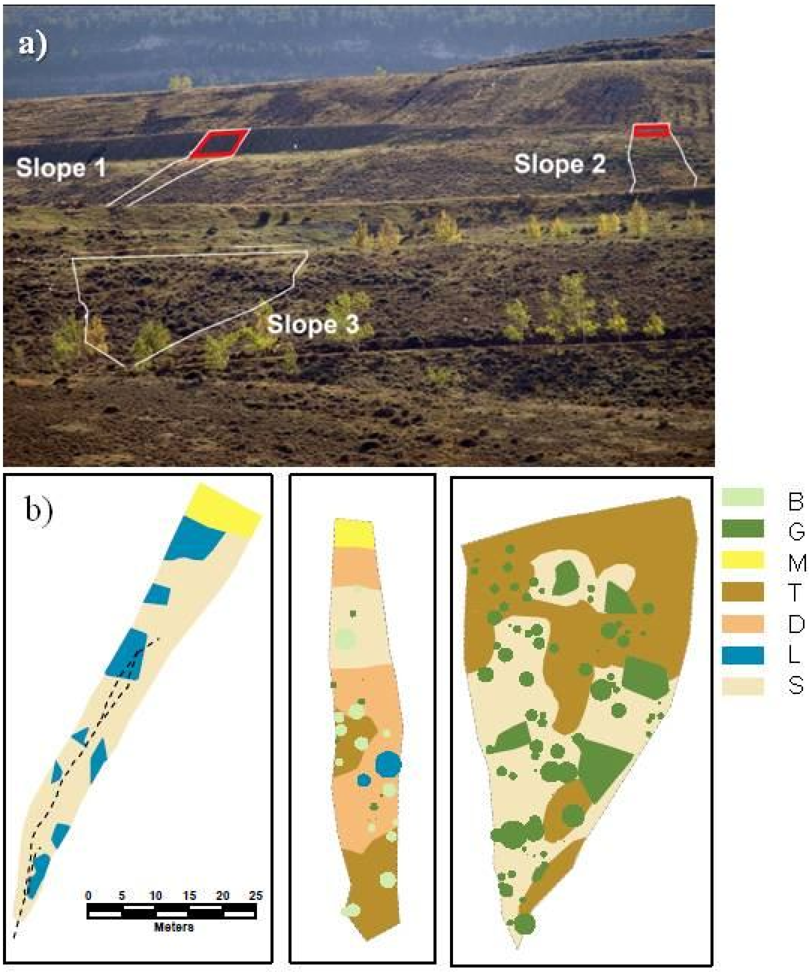


Figure 5.1. a) Selected slopes; the red line shows the upslope structure that generates extra-overland flow. b) abundance and spatial distribution of the different vegetation patches in the three slopes. B: *Brachypodium;* G: *Genista*; M: *Medicago;* T: *Thymus*; D: *Dactylis*; L: *Lolium*; S: *Santolina*; .

Table 5.1. Basic characteristics of the three experimental slopes.

	N	Slope 1		Slope 2		Slope 3	
Date of reclamation		1988		1988		1987	
Topography							
Slope area (m^2)		497.53		510.59		1474.30	
Slope gradient (º)		20		20		20	
Area of *Water-Contributing Area* (m^2)		50.36		22.67		0	
Aspect		North		North		North	
[1]*Soil traits*							
Stoniness (%)	9	39.17 ± 4.54	a	40.50 ± 3.16	a	41.91 ± 3.36	a
Sand (%)	9	44.75 ± 2.64	a	45.56 ± 2.68	a	43.95 ± 2.98	a
Silt (%)	9	28.68 ± 0.27	a	25.19 ± 1.07	a	29.75 ± 1.23	a
Clay (%)	9	26.57 ± 2.38	a	29.25 ± 1.61	a	26.30 ± 1.76	a
Texture	9	Clay loam		Clay loam		Clay loam	
pH -H_2O; w/v: ½-	9	8.38 ± 0.25	a	8.32 ± 0.15	a	8.01 ± 0.22	a
EC -w/v: ½- (dS m^{-1})	9	0.31 ± 0.06	a	0.31 ± 0.01	a	0.68 ± 0.42	a
Organic matter (%)	9	1.18 ± 0.25	a	1.55 ± 0.34	a	1.99 ± 0.37	a
$CaCO_3$ (%)	9	7.34 ± 0.44	a	6.17 ± 0.38	a	6.84 ± 0.28	a
[2]Bulk density (g cm^{-3})	27	1.48 ± 0.08	a	1.52 ± 0.03	a	1.43 ± 0.01	a
[3]*Cover features*							
Bare soil cover (%)	105	44.63 ± 3.06	a	32.29 ± 3.94	b	23.71 ± 2.75	b
Stone cover (%)	105	25.54 ± 2.97	a	22.69 ± 1.60	a	21.09 ± 2.57	a
Litter cover (%)	105	5.43 ± 1.81	a	1.09 ± 0.39	b	4.00 ± 1.87	ab
Vegetation cover (%)	105	24.40 ± 2.81	a	43.94 ± 4.07	b	51.20 ± 4.16	b
[3]*Plant traits*							
Species Richness	105	3.83 ± 0.28	a	6.43 ± 0.44	b	9.26 ± 0.55	c
Shannon’s index	105	0.80 ± 0.08	a	1.21 ± 0.08	b	1.30 ± 0.08	b
Hydrological features							
[4]Sheet Erosion Index	9	0.70 ± 0.09	a	0.59 ± 0.05	a	0.52 ± 0.06	a
[5]Rill density (m m^{-2})	3	0.58		0.00		0.00	
[6]Rill erosion rate[4] (t ha^{-1}yr^{-1})	3	8.41		0.00		0.00	
[7]Runoff coefficient (%)	17	14.52 ± 3.07	a	2.07 ± 0.51	b	0.32 ± 0.17	b
[7]Sediment yield (g m^{-2})	17	102.47 ± 36.49	a	4.51 ± 1.74	b	0.22 ± 0.07	c

Abbreviations: N: Number of samples; EC: Electrical conductivity; w/v: relation weigh (soil) / volume (water).

[1] Measured in three composite samples (each formed by three subsamples) from the first 10 cm in three regularly distributed transects along the gradient of the slope.

[2] Measured in nine randomly distributed unaltered soil cores (3 cm height by 5 cm diameter).

[3] Cover, visually estimated during spring 2006 in 35 regularly distributed 0,25 m^2 plots per slope.

[4] Measured by relation: stone cover/stoniness; following Moreno-de las Heras (2008).

[5] Linear rill length (m) measured per surface area (m^2).

[6] Measured from rill network dimensions following Morgan (1997).

[7] Measured in slope runoff plots from November 2007 to December 2008, tested using *post hoc* Wilcoxon-Nemenyi-McDonald-Thompson tests.

Values with the same letters (a-c) within rows do not differ significantly at α=0,05. Tested using Kruskal-Wallis and Mann-Whitney *post hoc* tests.

Field measurements

Vegetation and soil seed bank

In each vegetation patch, visual surveys of the presence of plant species were carried out in 15 randomly selected 50x50 cm quadrats to provide a fine description of vegetation features (indicator species and richness). Presence of plant species was considered rather than abundance because in some patches the existence of two layers of vegetation (canopy and understorey) could influence results. Total vegetation cover was estimated in 6 randomly selected 50x50 cm plots in each patch.

In September 2007, before the arrival of the autumn rains, soil samples (38.5 cm^2 area × 4 cm deep) were collected in order to analyze differences in floristic composition of the soil seed bank of the different vegetation patches. Ten random samples were collected from each patch. Each of these samples was subdivided into four subsamples that were placed in 250 ml plastic containers over a 5 cm vermiculite layer. The floristic composition of the soil seed bank was determined after germination under optimal conditions in a greenhouse.

Rainfall simulations

Three rainfall simulations were carried out in each vegetation patch in the summer of 2008. Experiments were carried out using a single nozzle (HARDI® 1553) rainfall simulator based on the model described by Cerdà *et al*. (1997b). Rainfall simulations were performed using a pressure of 2 kg cm^{-2} over 0.24 m^2 plots with the nozzle placed 2 m above the soil surface and protected from wind effects with plastic sheets. Calibration under these conditions resulted in the following rainfall characteristics: rainfall intensity was 75.2 mm h^{-1}; rainfall uniformity (*sensu* Christiansen 1942) was 72.94 %; drop diameter D_{50} (*sensu* Anderson 1948) was 1.7 mm and mean terminal speed was 4.0 m s^{-1}, which implies a kinetic energy of 13.4 J mm^{-1} m^{-2}. Twenty-one steel rings (55 cm diameter, 15 cm height) were installed (by inserting them 5 cm into the soil) in the seven vegetation patches (3 in each). Each of these plots had a 2.5 cm diameter drainpipe outlet to collect runoff and sediments. Rings were installed in October 2007 in order to ensure natural consolidation of soil after surface disruptions caused during ring installation. The duration of each rainfall event was variable (until runoff rate were stable, lasting a minimum of 30 min). Runoff discharge in each rainfall experiment was collected manually at 1 min intervals, from the beginning of runoff. The volume of each 1 min sample was measured and accumulated in a plastic bucket. After each rainfall experiment, runoff samples stored in the bucket were mixed and a homogeneous aliquot (0.5 l) was extracted. Sediment concentration (g l^{-1}) was determined in aliquots by weighing after oven drying (105ºC). Time to runoff (min) and runoff coefficient (%) were also measured. Finally, after each simulation, a vertical cut in the soil profile was used to measure the depth reached by the wetting front using a measuring tape (cm). This depth value was divided by the duration of the rainfall experiment to obtain a soil profile moistening rate (cm h^{-1}). To minimize the influence of water salinity on hydrological soil responses (Agassi *et al*. 1981) water with proper EC and SAR (EC=0.28dS m^{-1}, SAR<1) was used.

Soil moisture dynamics

To study the soil moisture dynamics in vegetation patches TDR (Time Domain Reflectometry) sensors were installed horizontally at different depths along the soil profile: 5, 25 and 50 cm, totalling four replicates in each vegetation patch. Soil moisture measurements were taken periodically (every 15 days without rain and within 5 days after each rainfall event) from April to December 2008. A TDR (Tektronix® 1502C) cable tester was used to collect the data, following the methodology proposed by Cassel *et al*. (1994), with an accuracy of 94 % in the determination of soil moisture.

Soil properties

Three composite soil samples (each sample formed by three homogeneously mixed subsamples, randomly distributed within each patch) were taken from the first 10 cm of the soil profile in each patch. Stoniness (%) was determined as the content of soil particles > 2 mm. General physicochemical characteristics were determined using standardized methods (MAPA 1994). A standard pressure chamber (Klute 1986) was used to determine soil water content (%, v/v) at four different pressures ranging from saturation to permanent wilting point (Ψ=0; -0.01, -0.03, -1.50 MPa). Parameterization of the soil water retention characteristic curves was made according to Van Genutchen (1980). Additionally, soil bulk density (g cm^{-3}) was determined in the upper part of the soil profile using fifteen unaltered soil cores (3 cm height x 5 cm diameter) collected in each vegetation patch, and soil surface strength was measured with a pocket penetrometer (Geotester®) in 15 randomly distributed samples per patch, four times per year (seasonal measurements).

Laboratory measurements

Seed germination under different water potentials

The hydrological requirements for germination in the eight most abundant species in slopes were studied (*G. scorpius, B. retusum, L. perenne, T. vulgaris, D. glomerata, S. chamaecyparissus, M. sativa* and *Aegilops geniculata*). For *M. sativa*, data obtained from a similar experiment carried out by our research group in the same study area (Moreno-de las Heras 2009) were used. Seeds of *S. chamaecyparissus* and *A. geniculata* were collected in the field. As field seed collection of the remaining species was impractical because of low production, they were obtained from two local seed suppliers (*Semillas Montaraz, S.A.* and *Zulueta Corporacion para la Naturaleza, S.A.*). *G. scorpius* seeds were scarified by submerging in sulphuric acid 95-98 % for 10 min following Bochet *et al*. (2007). Germination of seeds was studied under eight different water potentials (Ψ=0; -0.03; -0.10; -0.20; -0.33; -0.62; -1.10 and -1.50 MPa) representing a gradient of soil moisture between saturation and permanent wilting point. Water potentials were simulated using polyethylene glycol concentrations (PEG-6000) following the standard equations of Michel *et al*. (1983). Ten replicates per water potential were prepared. Replicates consisted of 15 seeds placed in a 9 cm diameter Petri dish, on a bed composed of a layer of hydrophilic cotton and filter paper. Replicates were moistened with 35 ml of distilled water (control; Ψ=0MPa) or PEG solutions (Ψ ranging from -0.03 to -1.50 MPa). To avoid water potential variations by evaporation, Petri dishes were sealed with PVC sheets. Petri dishes were placed in a phytotron under controlled conditions (day length= 12 h; air temperature= 20 ºC; relative air humidity= 75 %).

Germinations were monitored every three days; considering a germination when at least 2 mm of radicle emerged. The experiment lasted 46 days and then germination rate (%) of each species was calculated.

Data analyses

Vegetation and seed bank

The "characteristic species" of each vegetation patch was determined by Indicator Species Analysis (Dufrene & Legendre 1997). We performed ANOVAs and Tukey *post hocs* to test significant differences in species richness and Shannon's diversity (vegetation and seed bank) between vegetation patches. Kruskal Wallis and Mann Whitney *post hoc* tests were used to analyze differences soil seed bank density.

Rainfall simulations

Water infiltration was estimated as the balance between precipitation and measured runoff. Two descriptors of the infiltration process (final infiltration rate and shape coefficient of the apparent infiltration curve) were obtained by fitting, from minute-estimated infiltration values, the Horton-type equation proposed by Borselli *et al*. (1996).

$$I_t=I_f+(I_0-I_f)\exp^{-pt/K}$$

Where: I_t is the instantaneous infiltration rate (mm h^{-1}); I_f is the final steady infiltration rate (mm h^{-1}); I_0 is the initial infiltration rate (mm h^{-1}); p is the rainfall intensity (mm h^{-1}), t is the time (h) and K is a coefficient which describes the shape of the apparent infiltration curve.

Differences in response parameters (final infiltration rate, K shape coefficient, runoff coefficient, soil profile moistening rate and sediment concentration) between vegetation patches were analyzed by using ANOVAs and Tukey *post hoc* tests for those cases in which parametric assumptions were satisfied and Kruskal Wallis and Mann-Whitney *post hoc* tests for non-parametric data.

Soil moisture

We performed repeated measures ANOVA with soil water content data from vegetation patches, vegetation patch and sensor depth being the between-subject factors, and with time as the within-subject factor.

Soil water content was estimated at the slope scale (at 5, 25 and 50 cm soil depth) by considering the abundance (% surface cover) and soil moisture of each vegetation patch in each slope. A Friedman analysis with the Wilcoxon-Nemenyi-McDonald-Thompson *post hoc* test was performed to compare the three slopes.

Relationships between vegetation traits and soil hydrology

We performed linear fittings to study relationships between vegetation characteristics (species richness, diversity and density of soil seed banks) and hydrological traits (infiltration parameters and soil water content) in each vegetation patch.

Soil properties

Differences in soil bulk density between patches were tested by using ANOVA and Tukey´s *post hoc* tests. For soil surface strength, the Friedman and Wilcoxon-Nemenyi-McDonald-Thompson *post hoc* tests were used. The remaining soil physicochemical characteristics were tested using Kruskal-Wallis and Mann-Whitney U Test *post hoc* tests.

Seed germination

A sigmoid shape response function ($f= a/(1+\exp(-(x-x_0)/b))$) was fitted to the germination results obtained for each species (Ahmadi & Ardekani 2006). From this curve parameter "x_0" was used to estimate the water potential threshold value for seed germination and parameter "a" to estimate maximum germination rate. Threshold water potentials were transformed into soil moisture values by using the parameterized characteristic curves of soil water retention. The obtained soil moisture threshold values were used to calculate the number of days suitable for germination in each vegetation patch throughout the year: the days in which soil moisture at 5 cm depth was above the threshold value for each species were counted in each vegetation patch during the two seed germination pulses (spring and autumn). A general temperature threshold of 10 ºC (minimum temperature for seed germination) was also considered. A "germination suitability index" was estimated, calculated by multiplying the number of suitable days for germination and the maximum germination rate for each species in each vegetation patch. A non-parametric bi-factorial ANOVA was applied to these data, with species and vegetation patch as factors and the germination suitability index as the dependent variable.

At the slope scale, the "germination suitability index" was estimated considering the abundance (% cover) of each vegetation patch. A Friedman analysis and the *post hoc* Wilcoxon-Nemenyi-McDonald-Thompson test were applied to compare these values between the three slopes.

Indicator Species Analysis was performed with the PC-ORD package (McCune & Mefford 1999). We used the R program (R_Development_Core_Team 2009) for the non parametric bi-factorial ANOVA (using the "adonis" function of the "vegan" package). The *post hoc* Wilcoxon-Nemenyi-McDonald-Thompson test was performed with the "coin" and "multcomp" packages using the code of "Tal Galili", published in r-statistics.com (*http://www.r-statistics.com/2010/02/post-hoc-analysis-for-friedmans-test-r-code*). The remaining statistical analyses were performed using STATISTICA (Statsoft 2001). Data analyzed using ANOVA and Tukey tests ful filled parametric assumptions. The scientific names of the species are in accordance with *Flora Europaea* (Tutin *et al*. 1964-1980).

Results

Vegetation and seed bank

A total of 54 different species were identified. We found significant differences in species richness ($F_{6,98}$ =15.543, p<0.01; Fig. 5.2) and vegetation cover (Kruskal-Wallis test H=34.254, p<0.01; Table 5.2) between vegetation patches. The "characteristic species" of each vegetation patch obtained with Indicator Species Analysis are shown in table 5.2. Seed density (Kruskal-Wallis test H=36.155; p<0.01, species richness ($F_{7,72}$ = 9.855, p<0.01) and Shannon`s diversity ($F_{7,72}$ =5.734, p<0.01; Fig. 5.2) of soil seed banks also showed significant differences between vegetation patches (Fig. 5.2). Two vegetation patches, *Medicago* and *Genista*, represent the more extreme cases, with *Genista* patches having the highest values of vegetation cover, species richness and density of seeds. *Medicago* patches, on the contrary, had the lowest values.

Rainfall simulations

We found significant differences in final infiltration rates ($F_{6,14}$ =3.6957, p=0.02; Fig. 5.3a), soil profile moistening rates ($F_{6,14}$ =9.7074, p<0.01; Fig. 5.3c) and sediment concentrations (Kruskal-Wallis test H=17.097, p<0.01; Fig. 5.3d) between vegetation patches. We observed similar differences between patches to those for floristic traits, *Genista* and *Medicago* being the most different patches (Figs. 5.2 and 5.3).

Soil moisture measurements

The repeated measures ANOVA applied to soil moisture data showed significant effects of patch, sensor depth and time (see table 5.3). *Genista, Brachypodium, Lolium* and *Thymus* patches showed higher soil water content. Regarding sensor depth, soil moisture increased with depth and, with respect to time, it was higher during spring and autumn (Fig. 5.4). There was a significant interaction between

Table 5.2. Indicator species and vegetation cover of the seven vegetation patches.

Vegetation community	**Indicator species**	**Vegetation cover (%)**	
Genista	*Genista scorpius, Anacyclus clavatus, Bromus rubens, Cerastium pumilum, Eryngium campestre, Plantago lanceolata, Sanguisorba minor, Xeranthemum inapertum*	81.33 ± 12.51	a
Brachypodium	*Brachypodium retusum, Avena sterilis, Avenula bromoides, Barkhausia haenseleri, Festuca sp., Koeleria vallesiana*	93.67 ± 3.63	a
Lolium	*Lolium perenne*	67.33 ± 11.05	a
Thymus	*Thymus vulgaris, Desmazeria rigida*	23.33 ± 3.64	ab
Dactylis	*Dactylis glomerata*	17.33 ± 1.69	ab
Santolina	*Santolina chamaecyparissus*	19.33 ± 6.14	ab
Medicago	*Medicago sativa, Scorzonera laciniata*	2.67 ± 0.42	b

Table 5.3. Results of the repeated measures ANOVA applied to soil water content in vegetation patches.

Effect	**Degr. of freedom**	**F**	**p**
Vegetation patch	7	14.811	0.000000
Sensor depth	2	13.012	0.000031
Vegetation patch*Sensor depth	14	3.390	0.000795
Time	16	546.762	0.000000
Time*Vegetation patch	112	6.896	0.000000
Time*Sensor depth	32	18.608	0.000000
Time*Vegetation patch*Sensor depth	224	2.482	0.000000

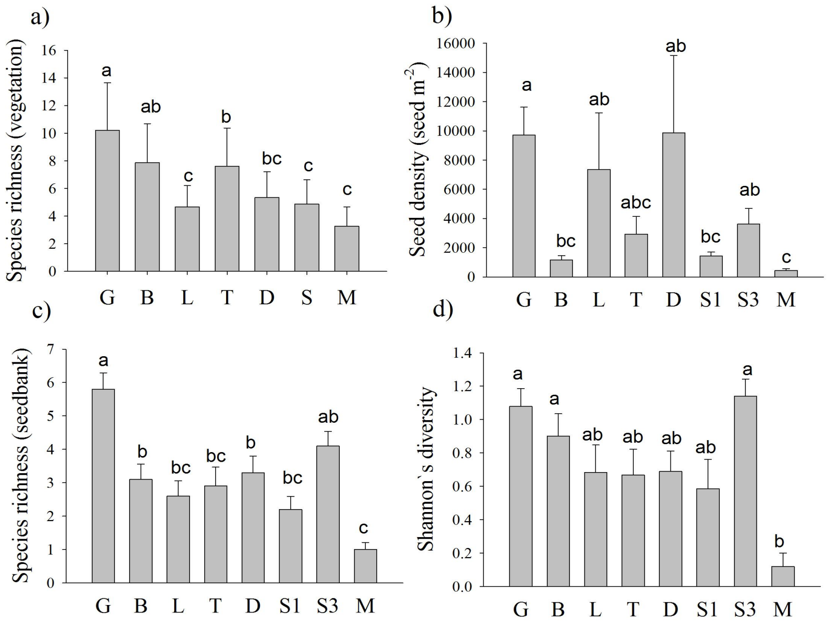


Figure 5.2. Floristic characteristics of the different vegetation patches. a) Plant species richness; b) Seed density of soil seed bank; c) Species richness of soil seed bank; d) Shannon's diversity of soil seed bank. G: *Genista*; B: *Brachypodium* L: *Lolium*; T: *Thymus*; D: *Dactylis*; S1: *Santolina* in slope 1; S3: *Santolina* in slope 3; M: *Medicago*. Different letters indicate significant differences ($p<0.01$).

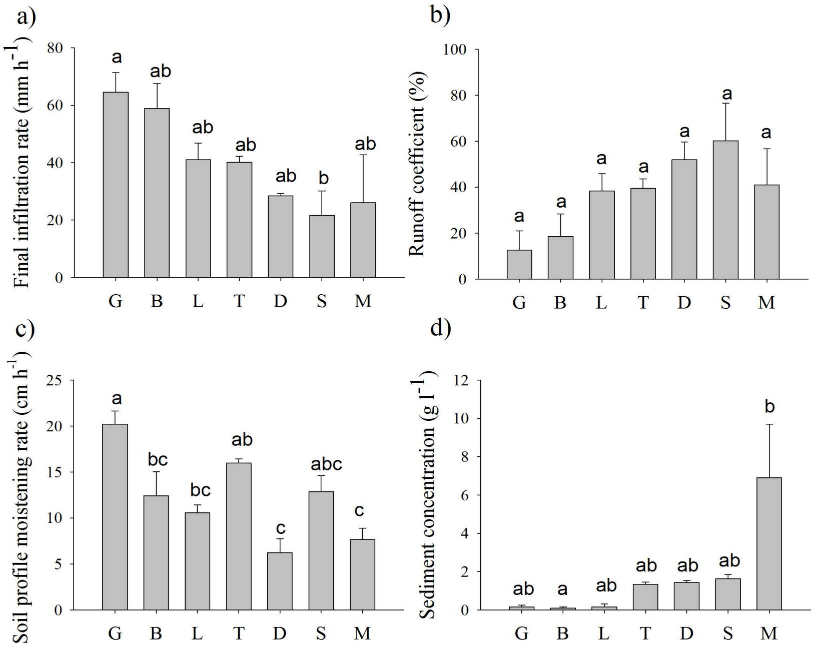


Figure 5.3. Soil hydrological properties of the different vegetation patches. a) Final infiltration rate (mm h^{-1}); b) runoff coefficient (%); c) soil profile moistening rate (cm h^{-1}); d) sediment concentration (g l^{-1}). G: *Genista*; B: *Brachypodium* L: *Lolium*; T: *Thymus*; D: *Dactylis*; S: *Santolina*; M: *Medicago*.

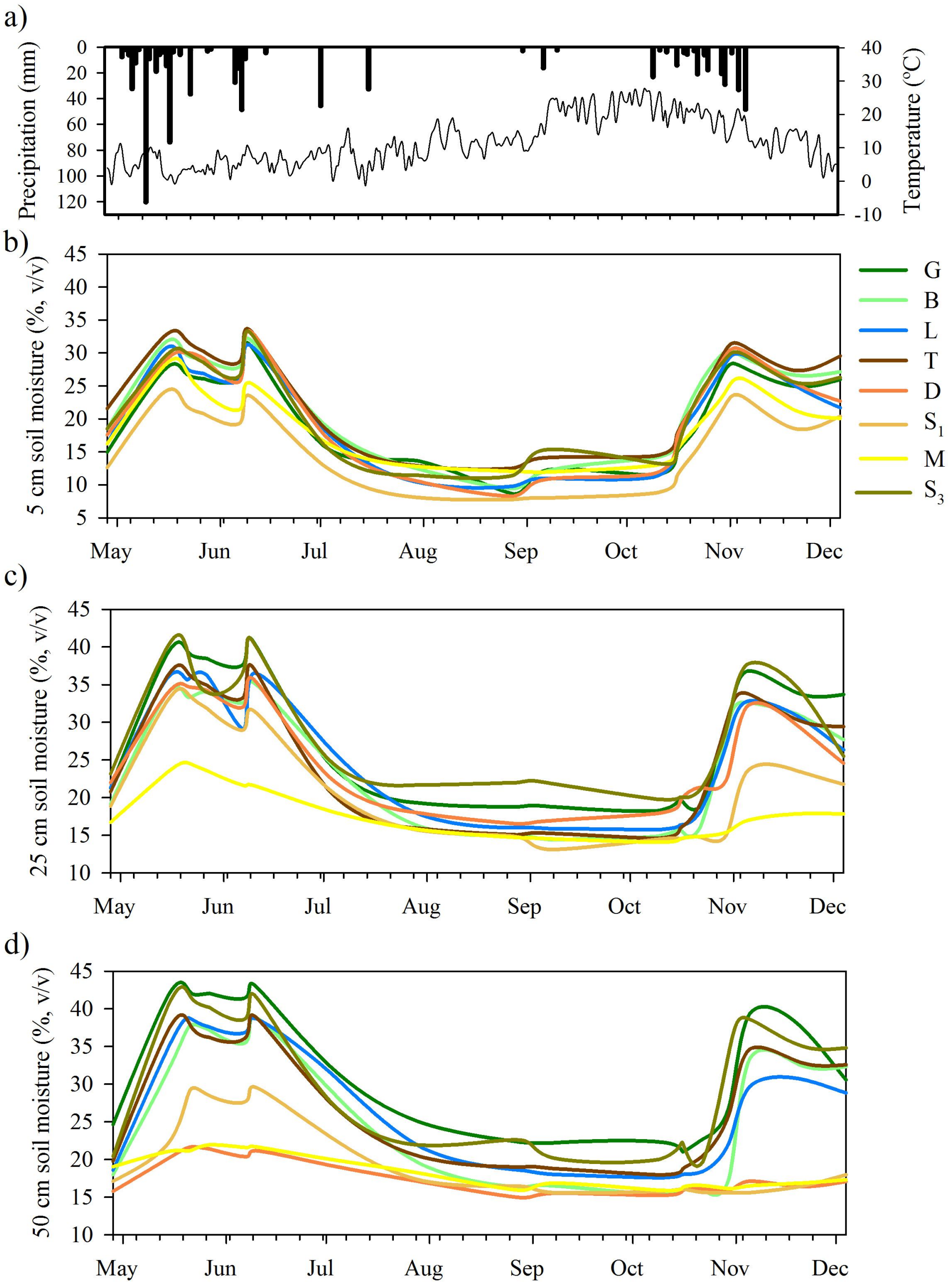


Figure 5.4. 2007-08 soil moisture dynamics in vegetation patches at three soil depths. b) 5cm; c) 25cm; d) 50 cm. G: *Genista*; B: *Brachypodium* L: *Lolium*; T: *Thymus*; D: *Dactylis*; S1: *Santolina* in slope 1; S3: *Santolina* in slope 3; M: *Medicago*. Figure 5.4a shows meteorological data (daily precipitation and daily mean air temperature) throughout the monitored period.

vegetation patch and sensor depth: soil water content in *Genista* patches increased with depth while *Brachypodium, Santolina* and *Medicago* patches showed constant values along the soil profile (Fig. 5.4). There was also a significant interaction between time and sensor depth, since differences in soil water content with depth became more patent during spring (Fig. 5.4). Detailed soil moisture dynamics of patches, interpolated along the soil profile during the study period, are shown in Appendix 5.A.

At slope scale we observed significant differences in soil moisture between slopes at the three soil depths (Friedman ANOVA p<0.001). Slope 3 showed the highest soil moisture values while slope 1 the lowest (Fig. 5.5).

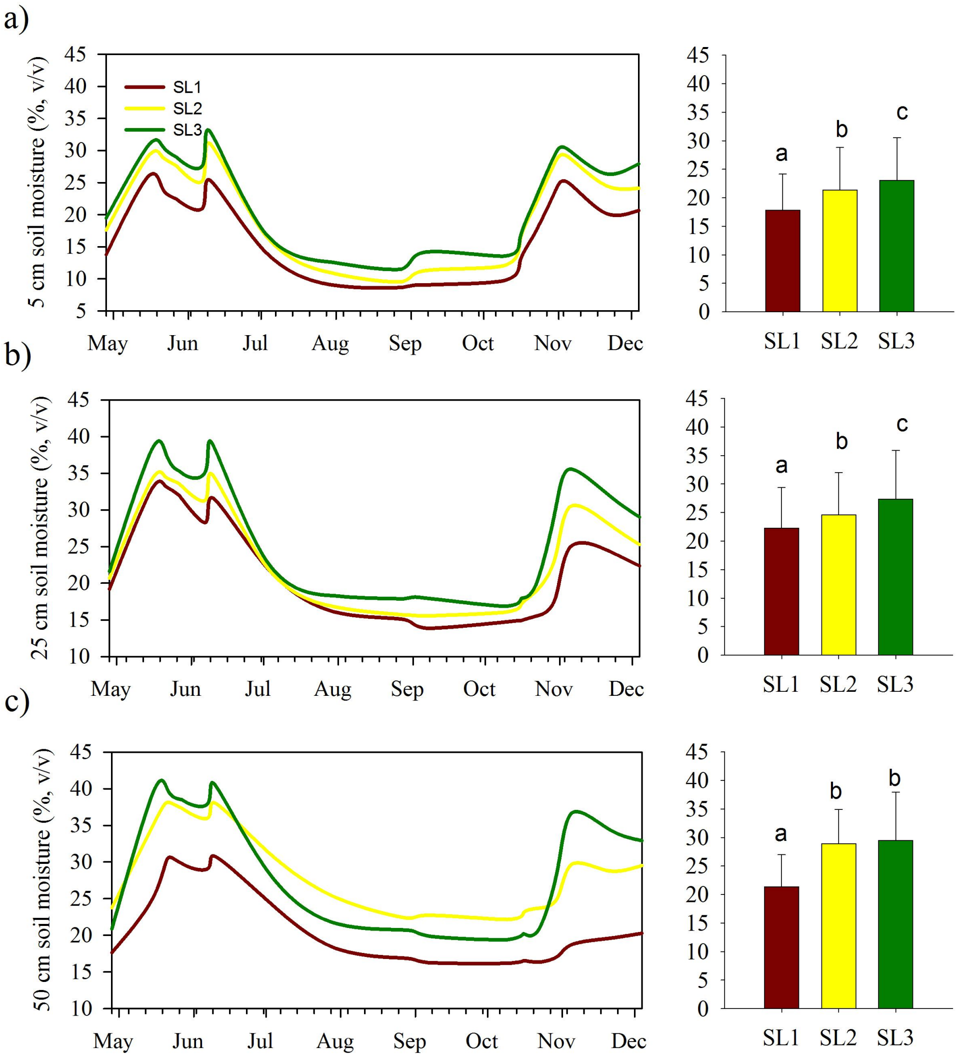


Figure 5.5. 2007-08 soil moisture dynamics and mean soil moisture values in the three slopes under study (SL1: slope 1, SL2: slope 2, SL3: slope 3). a) at 5 cm, b) at 25 cm, c) at 50 cm. Different letters indicate significant differences (p<0.01, *post hoc* tests).

Table 5.4. Results of linear fittings applied to hydrological and vegetation properties in vegetation patches. I_f: final infiltration rate; Qc: Runoff coefficient; Sm: Soil moisture. (**: p<0.01, *: p<0.05; n.s.: non significant).

Cover type features		I_f	Qc	Sm 5cm	Sm 25cm	Sm 50cm
Species richness (vegetation)	p	0.012	0.059	0.316	0.032	0.031
	R^2	0.746	0.567	0.199	0.635	0.643
		*	n.s.	n.s.	*	*
Vegetation cover	p	0.006	0.027	0.450	0.1087	0.052
	R^2	0.808	0.658	0.115	0.432	0.562
		**	*	n.s.	n.s.	n.s.
Species richness (seed bank)	p	0.048	0.132	0.555	0.009	0.088
	R^2	0.576	0.394	0.074	0.774	0.472
		*	n.s.	n.s.	**	n.s.
Shannon's diversity (seed bank)	p	0.032	0.155	0.452	0.003	0.069
	R^2	0.634	0.359	0.188	0.845	0.517
		*	n.s.	n.s.	**	n.s.
Soil seed bank density	p	0.529	0.709	0.669	0.068	0.557
	R^2	0.083	0.030	0.040	0.517	0.073
		n.s.	n.s.	n.s.	n.s.	n.s.

Relationships between vegetation traits and soil hydrology

Regarding plant species richness, we found a significant linear fit for final infiltration rate and soil moisture at 25 and 50 cm (Table 5.4). For vegetation cover, we found significant relationships with final infiltration rate and runoff coefficient. With respect to soil seed bank species richness and diversity we found significant relationships with final infiltration rate and soil moisture at 25 cm.

Soil properties

We detected significant differences in the main soil nutrients (N, P and organic matter Table 5.5), finding the highest and the lowest contents in *Genista* and *Medicago* patches respectively. There were also significant differences between patches in soil bulk density and soil surface strength, following the same trend as for soil nutrients. Maximum and minimum silt content values were found in *Medicago* and *Brachypodium* patches respectively (Table 5.5).

Seed germination

Germination experiments under controlled conditions highlighted a drastic effect of water potential on the germination rate of the different species (Fig. 5.6). Parameters "x_0", "a" and the R^2 of the fitted sigmoid equation together with soil moisture threshold values are shown in table 5.6. In general, graminoids had lower moisture requirements for germination when compared to woody species (*G. scorpius, T.vulgaris, S. chamaecyparissus*), except for *M. sativa*, which was introduced during revegetation practices. The non parametric bifactorial ANOVA applied to the germination suitability index showed significant effects of vegetation patch ($F_{7,192}$=13.643, p<0.001) and species ($F_{7,192}$= 6.986, p<0.001). Two species showed fewer suitable days for germination (*Genista scorpius* and *Brachypodium retusum*). With regards to vegetation patches, *Brachypodium* and *Genista* patches were more suitable microsites for germination, contrasting with *Santolina* patches in slope 1, which presents less suitable soil moisture values for germination (Fig. 5.7).

Table 5.5. Edaphic properties of vegetation patches.

	Genista	*Braquipodium*	*Lolium*	*Thymus*	*Dactilys*	*Santolina*	*Medicago*	p
								
pH^1	8.17±0.05 a	8.41±0.08 a	8.21±0.11 a	8.28±0.09 a	8.51±0.10 a	8.51±0.12 a	7.91±0.16 a	n.s.
EC^1 (dSm^{-1})	0.07±0.01 a	0.08±0.02 a	0.04±0.01 a	0.05±0.01 a	0.05±0.01 a	0.05±0.01 a	0.07±0.01 a	n.s.
Carbonates^1 (%)	8.00±0.76 ab	9.43±0.66 ab	10.10±0.42 ab	11.57±0.55 a	9.73±1.09 ab	9.50±1.07 ab	1.90±0.47 b	*
Nitrogen^1 (%)	0.18±0.04 a	0.12±0.01 ab	0.09±0.01 ab	0.09±0.01 ab	0.06±0.01 ab	0.06±0.01 ab	0.03±0.01 b	**
Organic matter^1 (%)	4.40±0.93 a	2.84±0.26 ab	2.10±0.16 ab	2.26±0.49 ab	1.57±0.17 ab	0.51±0.26 ab	0.31±0.07 b	**
C/N^1	13.90±1.74 a	13.83±0.32 a	14.10±0.68 a	15.30±2.91 a	15.77±0.84 a	5.83±2.94 a	6.87±1.82 a	*
Phosphorus^1 (%)	13.67±3.33 a	11.67±1.33 a	5.67±0.33 ab	3.67±0.33 ab	2.67±0.33 b	4.33±0.33 ab	4.33±0.33 ab	**
Clay^1 (%)	19.33±0.33 a	19.67±0.88 a	12.33±0.88 a	18.00±0.58 a	17.33±0.33 a	20.00±1.00 a	15.00±1.16 a	*
Silt^1 (%)	29.00±1.00 ab	25.67±3.18 a	38.33±6.12 ab	38.33±1.76 ab	43.00±2.00 ab	46.33±0.67 ab	52.67±0.88 b	*
Sand^1 (%)	51.67±1.21 a	51.33±3.93 a	49.33±6.64 a	43.67±1.21 a	39.67±2.33 a	33.67±0.33 a	32.33±2.03 a	*
AWC^1 (%)	6.40±0.28 a	8.52±0.29 a	8.07±1.44 a	6.92±0.11 a	9.01±0.41 a	8.80±0.20 a	9.09±0.66 a	n.s.
Bulk density^2 (g cm^{-3})	1.13±0.04 a	1.30±0.05 b	1.41±0.04 b	1.55±0.02 cd	1.41±0.03 bc	1.42±0.02 bc	1.61±0.01 d	**
Soil surface strength^3 (kg)^3	2.22±2.12 a	4.73±3.75 ab	4.53±4.01 ab	4.67±3.66 ab	4.76±3.87 ab	4.41±3.21 ab	5.39±3.39 b	*

Abbreviations: EC: Electrical conductivity; w/v: relation weigh (soil)/volume (water):1/2; AWC: Available water content. ^{1} Measured in three composite samples (each formed by three subsamples) in each vegetation patch or runoff contributing area from the first 10 cm. ^{2} Measured in 15 unaltered soil cores (3 cm height by 5 cm diameter); 3 randomly distributed soil cores for each 7 vegetation patch. ^{3} Measured in 15 randomly distributed samples in the different vegetation patches during four campaigns (seasonal measurements). Values with "*" differ significantly at α=0,05. Values with "**" differ significantly at α=0,01. ^{1} Tested using Kruskal-Wallis and Mann-Whitney U tests. ^{2}Tested using ANOVA and Tukey´s *post hoc* tests. ^{3}Tested using Friedman test and Wilcoxon-Nemenyi-McDonald-Thompson *post hoc* tests.

Table 5.6. Coefficients of the sigmoid function ($f=a/(1+\exp(-(x-x_0)/b))$) and corresponding water content at threshold x_0 for different species.

Species	x_0	a	R^2	Soil moisture (%)
Genista scorpius	-0.611	46.26	0.737	12.36 ± 2.47
Brachypodium retusum	-0.724	49.09	0.745	12.04 ± 2.46
Lolium perenne	-0.766	72.11	0.955	11.94 ± 2.46
Thymus vulgaris	-0.569	78.43	0.928	12.50 ± 2.47
Dactylis glomerata	-0.741	89.19	0.948	11.99 ± 2.46
Santolina chamaecyparissus	-0.620	81.20	0.874	12.34 ± 2.47
Medicago sativa	-0.416	88.93	0.970	13.13 ± 2.48
Aegilops geniculata	-1.161	79.33	0.796	11.19 ± 2.45

The germination suitability index was significantly different between slopes (Friedman ANOVA $p<0.001$, Fig. 5.8), conditions in slope 1 being less suitable for germination than in the other slopes.

Discussion

In our research we have explored vegetation-hydrology interactions in four steps, following the TTRP approach (Ludwig *et al*. 2005): a) the amount of soil water availability for plants in different vegetation patches; b) the enhanced infiltration activity of these different vegetation patches associated to soil water availability; c) the feedback effect of vegetation on soil water availability by increasing water infiltrability; and d) the modulation of ecohydrological interactions through the volume of overland flow running downslope, considered a key disturbance in these reclaimed systems (Moreno-de las Heras 2009).

Soil moisture availability in vegetation patches

Measured soil moisture values should be interpreted as a result of a balance between water inputs through infiltration and water use by plants, which results in, for example, higher hydric stress at the end of the drought period in patches with higher vegetation cover. The three vegetation patches that act as sinks (Merino-Martín *et al*. Chapter 4): *Lolium*, *Brachypodium* splays and *Genista* clumps, incorporated higher water volumes in depth which are consumed by vegetation, depleting soil moisture content at the end of the growth season (Fig. 5.4). The extra soil water content in these patches could derive from higher rainfall infiltration capacity on site and/or from the obstruction of overland flow exported from source patches. Overland flow, through the generation of different micro-geomorphologies (depositional micro-landforms), could be the main factor controlling water infiltration in *Lolium* and *Brachypodium* patches (field observations suggested that these patches were associated to runoff splays). The vegetation structure in *Genista* clumps (composed by a dense herbaceous understorey and the shrub canopy) is able to efficiently obstruct water flow coming from the upper part of the slope and thus is probably the main driving factor for infiltration. This interaction suggests a coupling between source-patches up-slope and downslope *Genista* deep-sinks.

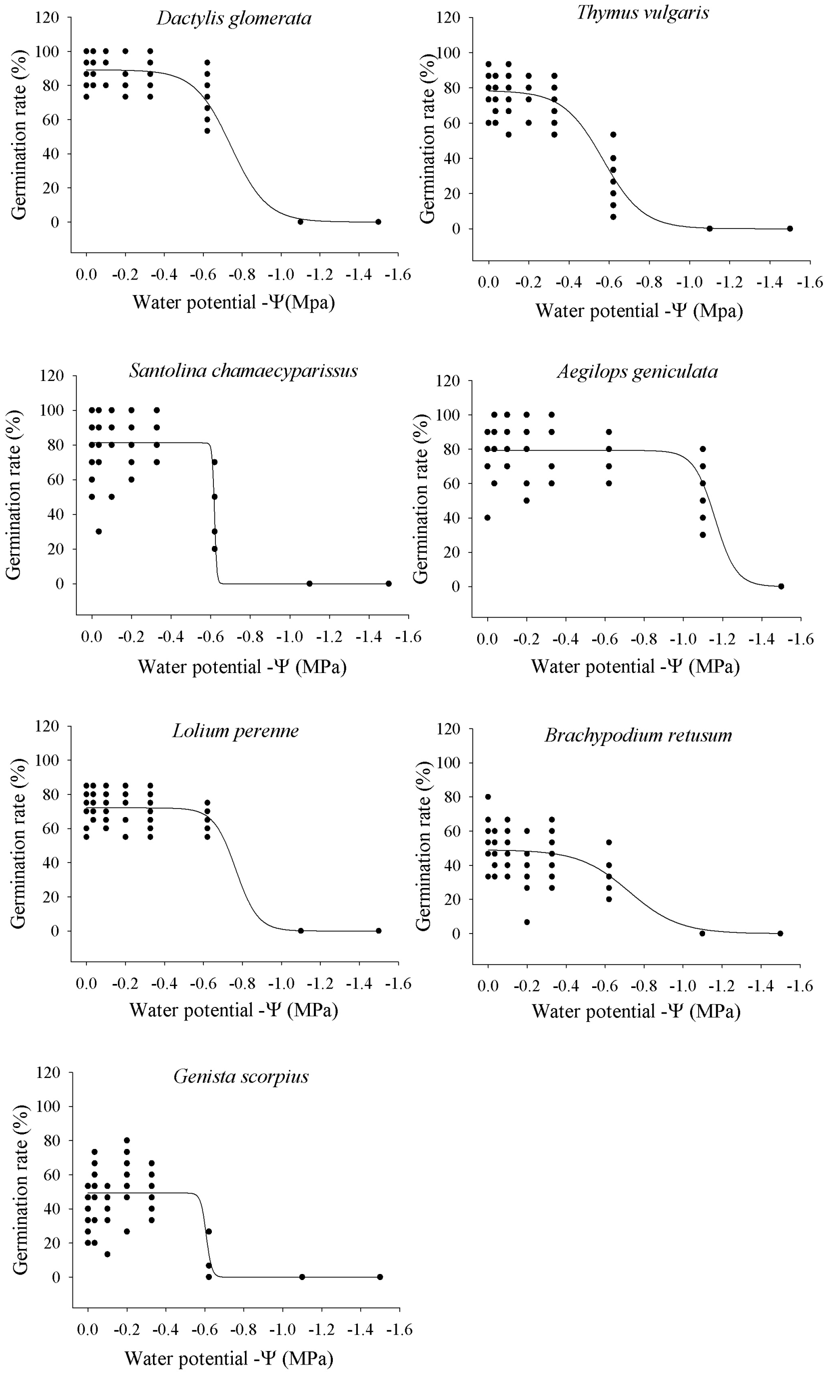


Figure 5.6. Seed germination rates of each species under different water potentials.

Ecological performance of vegetation patches associated to soil water availability

Sink patches (*Genista, Brachypodium, Lolium*) showed higher species richness and diversity compared with other patches located on the same slope (Fig. 5.2). We didn't find this pattern for soil seed bank density, where *Brachypodium* patches had a particularly low seed bank density. This could be explained by the resprouting strategy of *B. retusum*, advantageous over seeding, since it allows the species to persist continuously after disturbance (Bond & Midgley 2001).

Our findings highlight final infiltration rate and soil moisture in depth as the variables that best correlate with vegetation traits, in contrast with soil surface moisture, which was not correlated with ecological descriptors (Table 5.4). This suggests that soil moisture in depth is ecologically more significant than surface soil moisture. Focusing on these results, we performed a study of the implications of surface soil moisture for germination and we found that the different vegetation patches showed different suitable days for germination, with sink patches (*Genista* and *Brachypodium*) being the best micro-environments for germination (Fig. 5.7). Consequently, although surface soil moisture heterogeneity has implications for colonization; soil moisture in depth is more significant for subsequent vegetation establishment.

Plant community composition depends to a large extent on the amount and spatial distribution of soil moisture available for plants (Breshears & Barnes 1999). Our results showed how ecohydrological heterogeneity influences plant germination and colonization processes by driving the distribution of spatial soil moisture on slopes. In our study, we have not demonstrated a growth pulse in *Genista* patches (as expected under the TTRP conceptual framework), but we did find a pulse of floristic and soil seed bank richness in the understorey community.

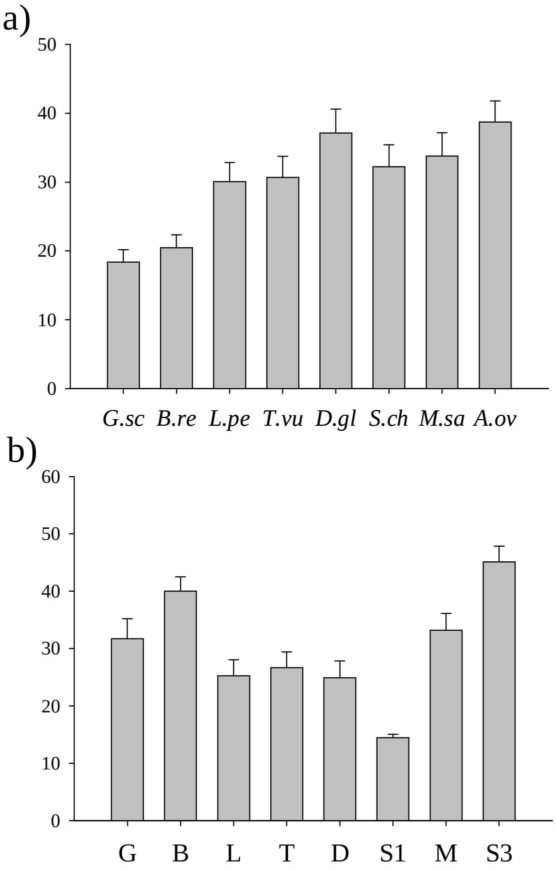


Figure 5.7. Germination suitability index for each species (a) and for each vegetation patch (b). G: *Genista*; B: *Brachypodium* L: *Lolium*; T: *Thymus*; D: *Dactylis*; S1: *Santolina* in slope 1; M: *Medicago;* S3: *Santolina* in slope 3). Species: G.sc: *G. scorpius*; B.re: *B. retusum*; L.pe: *L. perenne*; T.vu: *T. vulgaris*; D. gl: *D. glomerata*; S. ch: *S. chamaecyparissus*; M. sa: *M. sativa*; A. ge: *A. geniculata*.

Feedback effect of vegetation on soil water availability through and increase in water infiltration

A third type of vegetation-hydrology interaction that drives the performance of this semiarid community deals with the feedback effect of vegetation on soil water availability. Results from rainfall simulation experiments and soil properties analyses show that bulk density and infiltration capacity are higher in sink-patches than in the others, *Genista* patches being the most favourable for water infiltration and storage. Again, these differences can be attributed to both micro-landforms and vegetation in the case of *Lolium* and *Brachypodium* patches, but only to vegetation in the case of *Genista* patches, which allowed us to consider this interaction as a feedback effect.

This ecohydrological feedback can lead to "nucleation" processes, enhancing ecological and hydrological attributes for water collection and vegetation establishment. Rango *et al.* (2006) called these patches "islands of enhanced hydrologic activity" in the Chihuahuan Desert. Other authors refer to them as "fertility islands" (Cammeraat & Imeson 1998; Cerdà 1998; Puigdefábregas *et al.* 1999; Barthes & Roose 2002) .

The processes taking place in the studied slopes can be summarized as follows: initial conditions for water infiltration after slope reclamation were similar between the three slopes. However, they were soon modified by the effect of different overland flow routing along slopes, which generated erosion and deposition microsites in slope 1 and 2, but not in slope 3, where vegetation was established, directing the spatial redistribution of runoff. Therefore, two-way ecohydrological interrelationships were found: one two-way interrelationship where hydrology is the main driver (in slopes 1 and 2, where vegetation plays a passive role) and another two-way interrelationship (slope 3) where vegetation (*Genista* patches) is the main driver. Thus, *Genista scorpius* shrubs could be considered a key species for ecological succession in these semiarid reclaimed landscapes. However, as mentioned above, it should be taken into account that this patch appeared only in the slope subjected to low overland flow rates.

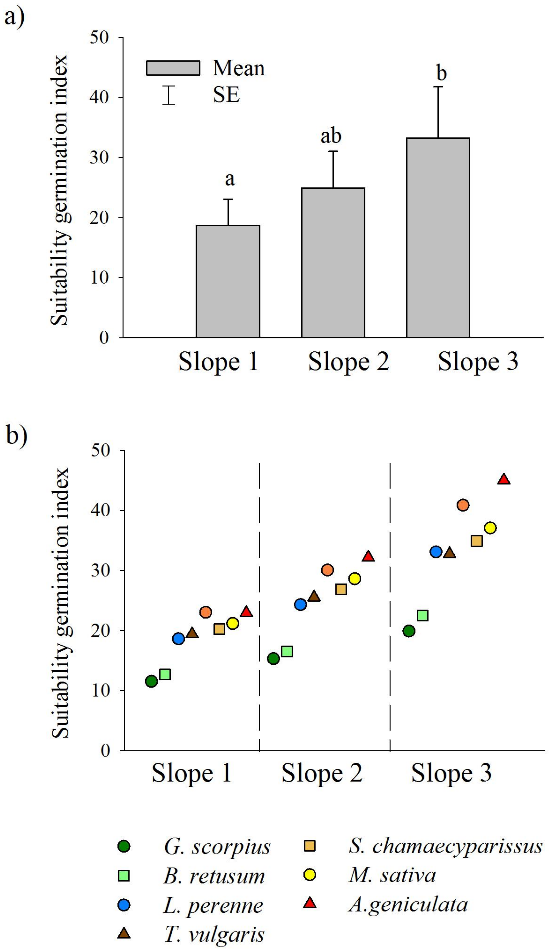


Figure 5.8. Germination suitability index in the three slopes for all species together (a) and for each species individually (b).

Similar ecohydrological interactions have been well described for semiarid ecosystems spatially structured in mosaics of coupled runoff sources and sinks resulting from feedbacks in a dynamic equilibrium (Valentin *et al*. 1999). In our case, we cannot anticipate wether our system is in a dynamic equilibrium through time or whether we are studying a temporary early stage of succession with a tendency to be covered by vegetation.

The modulation of ecohydrological interactions by overland flow volume.

Soil moisture content at the slope scale was related to overland flow (Fig. 5.5), showing higher soil moisture contents at the three depths whenever runoff volume decreased. The effects of erosion processes accentuating drought stress and decreasing productivity has been largely described (Lal 1998; Pimentel & Harvey 1999; Espigares *et al*. in press). Our study suggest that in reclaimed slopes where erosion rates are not particularly high, downslope overland flow is the main factor decreasing soil moisture at slope scale. We also found that overland flow at the slope scale decreased soil water content at the patch scale, as evidenced by soil moisture differences in *Santolina* patches between slopes 1 and 3, with significantly lower water contents in the slope with the highest runoff volumes. Hence, we propose that regardless of top-down (slope scale) or bottom-up (patch scale) processes dominate, if the capacity of the sink patch is exceeded, hydrological connectivity increases, leading to runoff export away from the slope and decreasing in-slope soil water content (the case of slope 1).

Overland flow routing through the slope also influences the germination opportunities for plants at the slope scale, decreasing germination probabilities as overland flow increases (Fig. 5.8). Traditionally, plant colonisation has been considered as limited by the distance between appropriate seed source areas and target areas in humid reclaimed environments (Kirmer & Mahn 2001; Novak & Prach 2003). Bochet *et al*. (2007) and Tormo *et al*. (2006), identified water availability as the main factor controlling germination processes in semiarid roadslopes. According to these results, we suggest that in reclaimed semiarid environments, where water is the main limiting factor, overland flow is one of the main factors driving plant colonization by increasing water deficit, not only for germination (at 5 cm), but also at 25 and 50 cm, for plant establishment and development.

Our results from semiarid reclaimed slopes were in general consistent with the TTRP conceptual model (Ludwig *et al*. 2005) demonstrating that vegetation patches have diverse hydrological behaviours and soil moisture contents, acting as runoff sources and sinks, with important ecological consequences derived from the heterogeneous spatial distribution of soil moisture. This general framework has been widely applied in natural semiarid ecosystems in dynamic equilibrium throughout the world; however, we have corroborated this framework for a highly dynamic and unstable restored ecosystem subjected to early stages of succession where it may only be taking place at this stage. Our results also agree with Imeson and Prinsen (2004), who stated that the spatial differentiation between source and sinks is a dynamic property of many semiarid ecosystems and it feeds back to plant growth as well as bare patch development.

The three selected slopes differed only in the up-slope generating areas, so we could say that our work is a natural experiment of the effects of overland flow on the ecohydrological interactions of reclaimed slopes. We also found an overland flow threshold (between slopes 2 and 3) that signals a change from a hydrological control to a vegetation control of ecohydrological interrelationships.

Rango *et al*. (2006) proposed that in order to remediate degraded rangelands, it may be appropriate to mimic nature by diverting water to target areas in order to create similar patterns to those of natural islands of hydrologically enhanced biotic productivity. Since we have shown that overland flow is an important driving force for microenvironmental patterns in these new reclaimed environments, it should be taken into account as one of the main factors in the design of restoration projects. These findings also highlight the importance of a correct design of reclaimed slopes, and the importance of an "expert management of runoff" if the desired objectives of a reclamation project are to be reached.

Acknowledgments

This work was supported by the Universidad de Alcalá, the project "Dinámica de la vegetación y de los procesos hidrológicos en laderas restauradas de la minería del carbón a cielo abierto en Teruel" funded by the Instituto de Estudios Turolenses and the project REMEDINAL (S2009AMB-1783), funded by the Regional Government of Madrid. We are grateful to the Utrillas Council for their active collaboration.We are also grateful to Carlos Martinez, Ignacio Morales, Enrique de la Montaña, Virginia Hernández, Pablo León, Carolina Morillo, Sara García and Bárbara Cuesta for their fieldwork help and Lucia Gálvez for language editing.

References

Agassi M., Shainberg I. & Morin J. (1981). Effect of Electrolyte Concentration and Soil Sodicity on Infiltration Rate and Crust Formation. Soil Science Society of America Journal, 45, 848-851.

Ahmadi S.H. & Ardekani J.N. (2006). The effect of water salinity on growth and physiological stages of eight Canola (Brassica napus) cultivars. Irrigation Science, 25, 11-20.

Anderson L.J. (1948). Drop-size distribution measurements in orographic rainfall. Bulletin of the American Meteorological Society, 29, 362-366.

Barthes B. & Roose E. (2002). Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. Catena, 47, 133-149.

Bochet E., Garcia-Fayos P., Alborch B. & Tormo J. (2007). Soil water availability effects on seed germination account for species segregation in semiarid roadslopes. Plant and Soil, 295, 179-191.

Bond W.J. & Midgley J.J. (2001). Ecology of sprouting in woody plants: the persistence niche. Trends in Ecology & Evolution, 16, 45-51.

Borselli L., Carnicelli S., Ferrari G.A., Pagliai M. & Lucamante G. (1996). Effects of gypsum on hydrological, mechanical and porosity properties of a kaolinitic crusting soil. Soil Technology, 9, 39-54.

Breshears D.D. & Barnes F.J. (1999). Interrelationships between plant functional types and soil moisture heterogeneity for semiarid landscapes within the grassland/forest continuum: a unified conceptual model. Landscape Ecology, 14, 465-478.

Calvo-Cases A., Boix-Fayos C. & Imeson A.C. (2003). Runoff generation, sediment movement and soil water behaviour on calcareous (limestone) slopes of some Mediterranean environments in southeast Spain. Geomorphology, 50, 269-291.

Cammeraat L.H. & Imeson A.C. (1998). Deriving indicators of soil degradation from soil aggregation studies in southeastern Spain and southern France. Geomorphology, 23, 307-321.

Cammeraat L.H. & Imeson A.C. (1999). The evolution and significance of soil-vegetation patterns following land abandonment and fire in Spain. Catena, 37, 107-127.

Cassel D.K., Kachanoski R.G. & Topp G.C. (1994). Practical considerations for using a TDR cable tester. Soil Technology, 7, 113-126.

Cerdà A. (1997a). The effect of patchy distribution of Stipa tenacissima L on runoff and erosion. Journal of Arid Environments, 36, 37-51.

Cerdà A. (1997b). Seasonal changes of the infiltration rates in a Mediterranean scrubland on limestone. Journal of Hydrology, 198, 209-225.

Cerdà A. (1998). Soil aggregate stability under different Mediterranean vegetation types. Catena, 32, 73-86.

Christiansen J.E. (1942). Irrigation by sprinkling. University of California Agricultural Experiment Station Bulletin, 670.

Dufrene M. & Legendre P. (1997). Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecological Monographs, 67, 345-366.

Espigares T., Heras M.M.-d.l. & Nicolau J.M. (in press). Performance of Vegetation in Reclaimed Slopes Affected by Soil Erosion. Restoration Ecology, doi: 10.1111/j.1526-100X.2009.00546.x.

Hancock G.R. & Willgoose G.R. (2004). An experimental and computer simulation study of erosion on a mine tailings dam wall. Earth Surface Processes and Landforms, 29, 457-475.

Imeson A.C. & Prinsen H.A.M. (2004). Vegetation patterns as biological indicators for identifying runoff and sediment source and sink areas for semi-arid landscapes in Spain. Agriculture Ecosystems & Environment, 104, 333-342.

Kirmer A. & Mahn E.G. (2001). Spontaneous and initiated succession on unvegetated slopes in the abandoned lignite-mining area of Goitsche, Germany. Applied Vegetation Science, 4, 19-27.

Klute A. (1986). Water retention: laboratory methods. In: Methods of soil analysis. Soil Science Society of America, Madison: USA, pp. 635-662.

Lal R. (1998). Soil erosion impact on agronomic productivity and environment quality. Critical Reviews in Plant Sciences, 17, 319-464.

Lavee H., Imeson A.C. & Sarah P. (1998). The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect. Land Degradation & Development, 9, 407-422.

Ludwig J., Tongway D., Freudenberger D., Noble J. & Hodgkinson K. (eds.) (1997). Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.

Ludwig J.A., Wiens J.A. & Tongway D.J. (2000). A scaling rule for landscape patches and how it applies to conserving soil resources in savannas. Ecosystems, 3, 84-97.

Ludwig J.A., Wilcox B.P., Breshears D.D., Tongway D.J. & Imeson A.C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology, 86, 288-297.

MAPA (1994). Métodos Oficiales de Análisis. In: Dirección General de Política Alimentária. Ministerio de Agricultura, Pesca y Alimentación (ed. Alimentación SGd) Madrid, Spain.

McCune B. & Mefford M.J. (1999). PC-ORD. Multivariate analysis of ecological data. In. MjM Software design Oregon, USA.

Merino-Martín L., Breshears D.D., Moreno-de las Heras M., Villegas J.C., Pérez-Domingo S., Espigares T. & Nicolau J.M. (Chapter 3). Ecohydrological Interrelationships Between Vegetation Patches and Soil Hydrological Properties Along a Disturbance Gradient: How Sources and Sinks of Runoff Determine a Restoration Threshold. Restoration Ecology. In review.

Merino-Martín L., Moreno-de las Heras M., Merino-Martín J.A., Pérez-Domingo S., Espigares T. & Nicolau J.M. (Chapter 4). Hydrological heterogeneity in mediterranean reclaimed slopes: runoff and sediment yield at the patch and slope scales along a gradient of overland flow. Hydrology and Earth System Sciences. In review.

Michel B.E. (1983). Evaluation of the water potentials of solutions of Polyethylene glycol-8000 both in the absence and presence of other solutes. Plant Physiology, 72, 66-70.

Moreno-de las Heras M. (2009). Efectos ecológicos de la erosión en laderas derivadas de la restauración de la minería a cielo abierto. In: Departamento de Ecología. Universidad de Alcalá Alcalá de Henares.

Moreno-de las Heras M., Merino-Martín L. & Nicolau J.M. (2009). Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. Catena, 77, 39-47.

Moreno-de las Heras M., Nicolau J.M. & Espigares T. (2008). Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecological Engineering, 34, 168-178.

Moreno-de las Heras M., Nicolau J.M., Merino-Martin L. & Wilcox B.P. (2010). Plot-scale effects on runoff and erosion along a slope degradation gradient. Water Resources Research, 46, 12.

Nicolau J.M. (2002). Runoff generation and routing on artificial slopes in a Mediterranean-continental environment: the Teruel coalfield, Spain. Hydrological Processes, 16, 631-647.

Novak J. & Prach K. (2003). Vegetation succession in basalt quarries: Pattern on a landscape scale. Applied Vegetation Science, 6, 111-116.

Papadakis J. (1966). Climates of the world and their agricultural potentialities, Buenos Aires.

Pimentel D. & Harvey C. (1999). Ecological effects of erosion. In: Ecosystems of disturbed ground (ed. Walker LR). Elsevier New York, pp. 123-135.

Puigdefábregas J. (2005). The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. Earth Surface Processes and Landforms, 30, 133-147.

Puigdefábregas J., Sole A., Gutierrez L., del Barrio G. & Boer M. (1999). Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in Southeast Spain. Earth-Science Reviews, 48, 39-70.

R_Development_Core_Team (2009). R: A language and environment for statistical computing. In. R Foundation for Statistical Computing Vienna, Austria.

Rango A., Tartowski S.L., Laliberte A., Wainwright J. & Parsons A. (2006). Islands of hydrologically enhanced biotic productivity in natural and managed arid ecosystems. Journal of Arid Environments, 65, 235-252.

Statsoft (2001). STATISTICA. Data Analysis Software System. In: Tulsa, OK.

Tongway D.J., Valentin C. & Seghieri J. (2001). Banded vegetation patterning in arid and semiarid environments: ecological processes and consequences for management. Springer-Verlag, New York, USA.

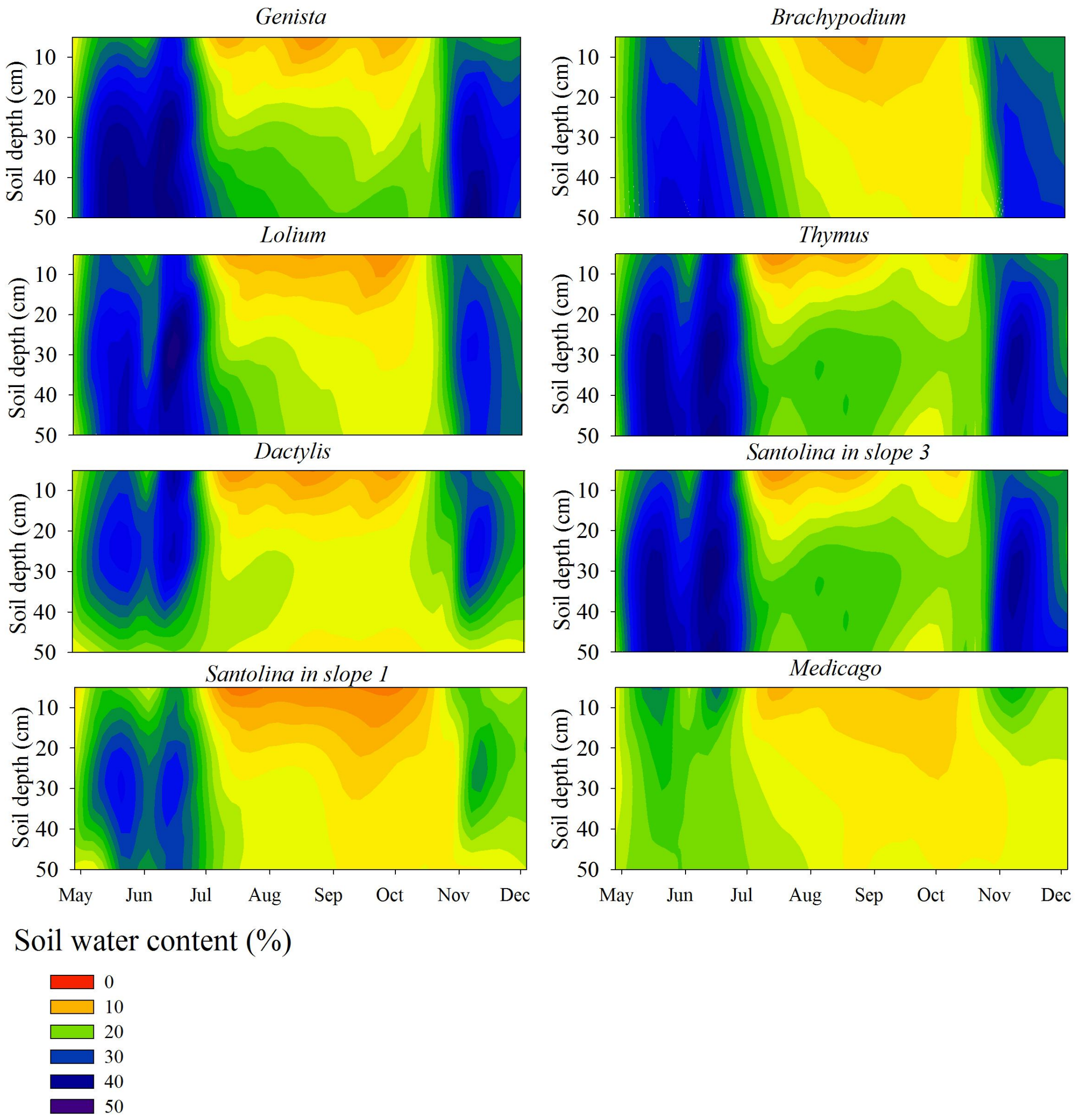
Tormo J., Bochet E. & García-Fayos P. (2006). Is seed availability enough to ensure colonization success?: An experimental study in road embankments. Ecological Engineering, 26, 224-230.

Tutin T.G., V.H. H., S.M. W. & D.A. W. (1964-1980). Flora Europaea. Cambridge University Press, Cambridge (UK).

Valentin C., d'Herbes J.M. & Poesen J. (1999). Soil and water components of banded vegetation patterns. Catena, 37, 1-24.

Van Genuchten M.T. (1980). A closed-form equation for predicting the hydraulic conductivity of unsaturated soils. Soil Science Society of America Journal, 44, 892-898.

Appendix 5.A. 2007-08 soil moisture dynamics in the vegetation patches along depth.



Capítulo 6

A Bárbara

Vale más mi sueño, que el dinero,
puedo vivir de una alegría,
de aquí payá colecciono recuerdos,
tu cuéntame cómo es tu vida.

Pa quí, pa llá, La Fuga

Capítulo 6

Interacciones ecohidrológicas en laderas artificiales en un gradiente de perturbación: sugerencias para la restauración ecológica en ambientes con restricciones hídricas

Este capítulo reproduce íntegramente el texto del siguiente manuscrito:

Tíscar Espigares; Luis Merino-Martín; Mariano Moreno-de las Heras y José Manuel Nicolau. En revisión. Ecohydrological interactions in artificial slopes along a disturbance gradient: insights for ecological restoration in drylands. Enviado a *Ecosystems*.

Resumen

Se realizó un experimento de campo para determinar la funcionalidad de las interacciones ecohidrológicas entre manchas de vegetación en laderas artificiales procedentes de minería de carbón a cielo abierto del centro-este de España. Se analizaron los efectos de la interrupción del flujo de escorrentía entre manchas sobre el contenido de humedad del suelo, el potencial hídrico de las especies leñosas, la biomasa herbácea y el banco de semillas en tres laderas sometidas a diferente grado de perturbación por escorrentía. Se analizó también la existencia de mecanismos de facilitación entre plantas en la ladera menos perturbada. El potencial hídrico de las especies leñosas, la biomasa de herbáceas y la heterogeneidad del banco de semillas fueron seriamente afectadas por la exclusión de escorrentía, siendo mayor el efecto según aumentó el grado de perturbación. Una relación positiva entre el matorral *Genista scorpius* y la vegetación herbácea fue encontrada en la ladera menos perturbada, posiblemente relacionada con la mejora del recurso hídrico bajo el dosel del matorral. Los resultados revelan la importancia de la escorrentía circulante por la ladera en los procesos de redistribución de escorrentía entre manchas de vegetación mediante la creación de islas de fertilidad que mejoran el rendimiento de la vegetación y con ello, el favorecimiento del cambio del control abiótico al control biótico del agua durante la sucesión de estos "nuevos ecosistemas". Por ello, se considera que las prácticas de restauración en ambientes secos obtendrían resultados significativamente mejores si se adoptara una estrategia de "manejo experto de la escorrentía".

Palabras clave: mediterráneo, restauración de minería, nuevos ecosistemas, interrupción de escorrentía, potencial hídrico, facilitación, banco de semillas, biomasa.

Ecohydrological interactions in artificial slopes along a disturbance gradient: insights for ecological restoration in drylands

Espigares, Tíscar[1]; Merino-Martín, Luis[1]; Moreno-de las Heras, Mariano[2]
and Nicolau, José-Manuel[3]

[1]Departamento de Ecología, Universidad de Alcalá. Alcalá de Henares, Madrid, España.
[2]Faculty of Engineering and Built Environment, University of Newcastle, Callaghan, Australia.
[3] Departamento de Agricultura y Economía agraria, Escuela Politécnica, Universidad de Zaragoza, Huesca, España.

Abstract

We have conducted a field experiment to ascertain the role of ecohydrological interactions between vegetation patches in the ecological succession of artificial slopes derived from open cast coal mining in central-eastern Spain. We analyzed the effects of runoff interruption between patches on the leaf water potential of woody species, on the herbaceous biomass and on the soil seed banks in three slopes subjected to a different disturbance degree. We also looked for the existence of facilitation mechanisms between plants in the less disturbed slope. Leaf water potential of woody species, herbaceous biomass and heterogeneity of soil seed banks were seriously affected by runoff exclusion, and this effect was more intense as level of disturbance increased. A positive relationship between the shrub *Genista scorpius* and the herbaceous vegetation was found in the less degraded slope, possibly related to the improvement of water resources under the canopy of the shrub. The results point out the importance of runoff redistribution between vegetation patches in the evolution of artificial slopes by creating fertility islands that improve the performance of vegetation and by favoring the shift from an abiotic to a biotic control of water in these "novel ecosystems". Restoration practices in drylands may thus significantly improve if a "runoff expert management" strategy is adopted.

Keywords: Mediterranean, mining restoration, novel ecosystems, runoff interruption, leaf water potential, facilitation, soil seedbanks, biomass.

Introduction

Ecohydrological interactions constitute a key process to understand how ecosystems work, especially in drylands where water is the main limiting factor for biological productivity (Aguiar and Sala 1999; Porporato and Rodríguez-Iturbe 2002). In the last years many studies have highlighted the importance of runoff redistribution for the dynamics of arid and semiarid ecosystems worldwide, with examples from Africa (White 1970; Seghieri *et al.* 1997), America (Cornet *et al*. 1992; Reid *et al.* 1999; Bhark and Small 2003), Asia (White 1969), Australia (Dunkerley and Brown 1995; Ludwig and Tongway 1995) and Europe (Puigdefábregas *et al.* 1999; Calvo-Cases *et al.* 2003; Imeson and Prinsen 2004). These studies point out that spatial pattern of vegetation produces a mosaic of patches in which runoff source areas (bare soil patches) are more or less coupled with runoff sink areas (vegetated patches) that may benefit from the extra water supply coming from upslope. The Trigger-Transfer-Reserve-Pulse model proposed by Ludwig *et al.* (2005) is a useful framework that explains the role of spatial redistribution of water in drylands and its impact on vegetation growth. This mechanism not only improves the water efficiency of the ecosystem (Noy-Meir 1973) but also increases its stability, as Van de Koppel and Rietkerk (2004) have stated by showing that spatial interactions between vegetation patches (in terms of water flow) confer resilience and increase the adaptive capacity of arid ecosystems.

Spatial distribution of vegetation has been identified as an influential factor for the ecohydrological interactions in semiarid ecosystems, even more decisive than vegetation cover *per se* (Wilcox *et al.* 2003; Bautista *et al.* 2007), as connectivity between runoff source patches may create leaky systems where water and soil resources easily flow out. In this sense, some authors have also suggested the use of the patch-size distribution of vegetation as an indicator of ecosystem degradation (Kéfi *et al.* 2007; Scanlon *et al.* 2007). It is no wonder that ecohydrology constitutes a major challenge for the international environmental agenda, especially for the rehabilitation of degraded arid and semiarid ecosystems (Hannah *et al.* 2007). Nevertheless, while the degradation process has been widely studied, our knowledge about the ecohydrology of the opposite phenomenon, the "agradation" process, in semiarid areas is very scarce, especially for restored and artificial ecosystems. Cammeraat and Imeson (1999) studied the natural recovery of degraded areas of the Mediterranean Spain and concluded that vegetation patterns play a major role preventing water loss at the end of the slopes. However, we need a better understanding of the mechanisms involved in the ecohydrological interactions during the agradation process to improve our management of restored ecosystems and the design of restoration projects (Newman *et al.* 2006; Wilcox and Thurow 2006).

Mining reclaimed landscapes can be included in the group of "novel" or "emerging" ecosystems whose functioning has been scarcely studied yet (Hobbs *et al.* 2006, 2009) and they constitute a great challenge for ecological restoration in drylands, as many restoration projects yield unsuccessful results (Moreno-de las Heras *et al.* 2008). Usually, these artificial ecosystems are characterized by poor soils with low infiltration capacity and scarce nutrients that make its vegetal colonization difficult, even more when climate imposes severe water limitations (Nicolau and Asensio

2000). In these conditions, when vegetation cover is low, bare soil surfaces produce large amounts of runoff that usually trigger soil erosion processes governed by non linear dynamics (Moreno-de las Heras *et al.* 2009). Also, wrong topographic designs may result in convex and steep water contributing areas upslope and in runon inputs that intensify soil erosion processes with the development of rill networks that evacuate water from the slopes very efficiently and drive the system to a highly degraded state in which costly efforts must be applied for its rehabilitation (Espigares *et al.* in press).

Lavee *et al.* (1998) described a shift in the ecohydrological behavior of ecosystems along a climatic gradient: from arid systems with an abiotic control of water (runoff dominated) to Mediterranean ones in which biotic processes favor infiltration. Between both extremes, semiarid ecosystems are described as a mosaic-like pattern containing arid-dry patches that produce runoff and humid-wet patches that receive it as runon. Mediterranean constructed ecosystems can be classified as "semiarid azonal", as they often have to face such harsh conditions that they behave as if they were under semiarid conditions. Therefore, restoration efforts must drive the system to achieve a biotic control of water.

Ecohydrological interactions are complex processes since they are influenced by many crosscutting issues: spatial and temporal scales and environmental gradients (Newman *et al* 2006; Turnbull *et al.* 2008). In arid and semiarid environments the complexity increases as recent studies have shown that even the effect of scale varies with the degree of degradation (Wilcox *et al.* 2003; Moreno-de las Heras *et al.* 2010). At the same time, classical models that explain the interactions between plants under stress gradients seem to need a reappraisal, as different responses from those predicted are found in highly water stressed environments (Maestre *et al.* 2005). More experimental research is then needed to unravel the complex ecohydrological interactions between the biotic and the abiotic component of the ecosystems.

The main objective of this investigation is to analyze the ecohydrological interactions in constructed slopes derived from mining reclamation in Spain along a disturbance gradient and their implications for ecological succession of these novel ecosystems. The disturbance gradient in the experiment is represented by different runoff volumes of the slopes, as increasing runoff is generally accepted as a signal of degradation in semiarid ecosystems (Ludwig *et al.* 2005). Specifically, we analyzed the effect of the spatial redistribution of runoff between vegetation patches on the performance of plant communities and how this effect changed with different levels of runoff. For this purpose we have conducted an experiment of runoff exclusion under field conditions that allowed us to explore the ecological effects of the spatial redistribution of water through the answer of the following questions: a) What is the contribution of the spatial redistribution of runoff to the increase of soil moisture in the runoff sink patches? b) What is the contribution of the spatial redistribution of runoff to improve the performance of vegetation in the runoff sink patches? and c) How do these processes vary along a runoff gradient?

Our hypothesis is that spatial redistribution of runoff will benefit patches with more vegetation cover that receive water as runon from bare patches upslope, so we expect a reduction in soil water and a worsening of plant performance in the patches subjected to runoff exclusion. At the same time, we expect minor ecological effects of the spatial redistribution of runoff in the slopes that have to cope with the routing of low runoff volumes, as the higher total vegetation cover in these less degraded slopes would enhance soil properties and the biotic control of water through the appearance of facilitation mechanisms. In this sense, another objective of our investigation is to look for the existence of positive interactions between plants in the less degraded slope that would indicate a transition from an abiotic to a biotic control of water as ecological succession proceeds in the constructed slopes.

Our investigation provides empirical evidence of the importance of ecohydrological interactions during the agradation process of artificial ecosystems and also it delves into the scarce knowledge of the functioning of this new but widely spread man-made ecosystems. Some practical implications for the restoration in drylands are also discussed.

Materials and methods

Study area

This study has been carried out in the *El Moral* reclaimed mine (50 ha), located within the *Utrillas* coalfield experimental site, in central-eastern Spain (40º47'50"N, 0º50'26"W), with a height of approximately 1100 m a.s.l. The climate in the area is Mediterranean-Continental, with a mean annual temperature of 14 ºC (ranging from 6.7 ºC in December and 23.1 ºC in July). The local moisture regime is Mediterranean-dry according to Papadakis (1966), with a mean annual precipitation of 466 mm (mainly concentrated in spring and autumn) and a potential evapotranspiration of 759 mm. Thus, the main constraints for the biological activity in the area are a long frost period (from October to April) and the intense summer drought (from June to October).

The edaphic characteristics of the area were determined by the reclamation operations performed by the mining company *Minas y Ferrocarril de Utrillas S.A.* between 1987 and 1988, consisting in the construction of artificial hillslopes by means of covering the spoil bank with a layer of 100-250 cm of overburden substratum from the *Escucha* and *Utrillas* cretacic formations of *Albian* age (a non-saline and clay-loam textured spoil). Afterwards, the artificial slopes were revegetated by sowing cross-slope a mixture of perennial grasses (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis* and *Lolium perenne*) and leguminous herbs (*Medicago sativa* and *Onobrychis viciifolia*). Nowadays, the reclaimed area is composed of a set of hillslopes that have evolved very differently from the same original starting point, mainly driven by the effects of diverse amounts of overland flow derived from contributing areas upslope (channels and berms that produce runon, or bare steep banks at the top of the slope) (Moreno-de las Heras *et al.* 2008). In some cases, intense soil erosion processes have provoked rill networks that limit the development of soil and vegetation (Moreno-de las Heras 2009; Espigares *et al.* in press), while in others the constructed hillslopes are being colonized by woody species. For our study we have selected three artificial slopes subjected to different runoff volumes (slope runoff coefficients ranging from 0.5 to 11.2% for a total precipitation of 550.7mm during the study period). All hillslopes were North-faced and had a similar angle of 20º.

In each of the selected slopes we have identified two types of patches depending on their vegetation cover: the vegetation patches, those with the maximum vegetation cover (more than 65%) which we assume to act as runoff sink patches, and the inter-patches that have a very low vegetation cover (less than 20%), which we assume to act as runoff source patches, as a strong relationship between vegetation cover an runoff coefficient has been found in the area configuring a sharp hydrological control threshold between 30 and 50% cover (Moreno-de las Heras *et al.* 2009). Table 6.1 offers a general description of the main edaphic features of each patch in each slope. In slope 1, subjected to the highest runoff volume, vegetation patches consisted of tussocks of *L. perenne* with scattered individuals of *Santolina chamaecyparissus*, with a mean cover of 67%. In slope 2 vegetation patches consisted of dense tussocks of *Brachypodium retusum* with a cover of 93%; and in slope 3, with the minimum runoff volume, vegetation patches are formed by shrubs of the leguminous *Genista scorpius*, with a mean cover of 81%. Inter-patches consisted of bare soil areas with some scattered individuals of the chamaephytes *S. chamaecyparissus* and *Thymus vulgaris*, and small spots of *Dactylis glomerata* and *M. sativa*. An herbaceous community of annual plants also accompanied both type of patches.

Table 6.1. General features of the three slopes under study.

	Slope 1		**Slope 2**		**Slope 3**	
Total vegetation cover (%)	24.40 ± 2.81		43.94 ± 4.07		51.20 ± 4.16	
Vegetation patch cover (%)	21.79		10.69		18.18	
Inter-patch cover (%)	78.21		89.31		81.82	
Runoff coefficient^1 *(%)*	11.17		2.04		0.50	
	Vegetation patch	Inter-patch	Vegetation patch	Inter-patch	Vegetation patch	Inter-patch
pH^2 *-H*$_2$*O;W/v:1/2-*	8.21 ± 0.11	8.48 ± 0.15	8.41 ± 0.08	8.33 ± 0.03	8.17 ± 0.05	8.28 ± 0.17
EC^2 *-w/v:1/2-(dSm*$^{-1}$*)*	0.04 ± 0.00	0.05 ± 0.00	0.08 ± 0.02	0.05 ± 0.01	0.07 ± 0.01	0.06 ± 0.01
Carbonates^2 *(%)*	10.10 ± 0.42	7.60 ± 2.96	9.43 ± 0.66	8.90 ± 0.90	8.00 ± 0.76	9.03± 0.64
Total Nitrogen^2 *(%)*	0.09 ± 0.01	0.05 ± 0.01*	0.12 ± 0.01	0.06 ± 0.01*	0.18 ± 0.04	0.12 ± 0.01
Organic matter ^{2} *(%)*	2.10 ± 0.16	0.31 ± 0.21*	2.84 ± 0.26	1.67 ± 0.65	4.31 ± 1.01	1.93 ± 0.57
C/N^2	14.10 ± 0.68	4.40 ± 2.36*	13.83 ± 0.32	14.90 ± 2.99	13.90 ± 1.73	8.76± 1.63
Total Phosphorus^2 *(%)*	5.67 ± 0.33	4.33 ± 0.33	11.67 ± 1.33	3.67 ± 0.33*	13.67 ± 3.33	8.67 ± 2.18
Clay^2 *(%)*	12.33 ± 0.88	18.67 ± 2.03*	19.67 ± 0.88	17.67 ± 0.67	19.33 ± 0.33	20.00± 0.58
Silt^2 *(%)*	38.33 ± 6.12	48.33 ± 2.40	25.67 ± 3.18	43.00 ± 2.00*	29.00 ± 1.00	32.33 ± 0.67*
Sand^1 *(%)*	49.33 ± 6.64	33.00 ± 0.58*	51.33 ± 3.93	39.33 ± 2.67*	51.67 ± 1.20	47.67 ± 0.88*
AWC (%)	8.07 ± 1.44	9.31 ± 0.52	8.52 ± 0.28	8.42 ± 0.85	6.40 ± 0.28	6.09 ± 0.48
Bulk density^3 *(g cm*$^{-3}$*)*	1.40 ± 0.03	1.51 ± 0.04	1.30 ± 0.05	1.48 ± 0.05*	1.13 ± 0.04	1.49 ± 0.02**
Soil surface strength^4 *(kg)*	4.53 ± 0.13	4.85 ± 0.11	5.05 ± 0.35	4.81 ± 0.09	2.22 ± 0.16	4.29 ± 0.11**

Abbreviations: EC: Electrical conductivity; w/v: relation weigh (soil)/volume (water); AWC: Available water content.

[1] Measured during the study period, by installing a collector at the foot of each slope where runoff was channeled through a cemented outlet.

[2] Measured in three composited samples (each formed of three subsamples) in each vegetation patch or runoff contributing area from the first 10 cm.

[3] Measured in 15 unaltered soil cores (3 cm height by 5 cm diameter) randomly distributed per vegetation patch and inter-patch.

[4] Measured in 15 samples randomly distributed per vegetation patch or runoff contributing area.

Values with * differ significantly at α=0.05. Values with ** differ significantly at α=0.01. Tested using Mann-Whitney U Test.

Experimental design and field measurements

In each slope we identified ten couples of adjacent patches (a vegetation patch with an inter-patch immediately upslope). Each couple was selected so that all patches were of similar sizes. In half of the couples of each slope runoff flow between both patches was interrupted by inserting a steel sheet (50 cm high). Measurements of soil water content, leaf water potential of woody species, herbaceous biomass and floristic composition of the soil seed bank were taken in each vegetation patch (5 control and 5 subjected to runoff exclusion in each slope) to ascertain the ecological impacts of runoff exclusion. At the same time, in order to explore the existence of facilitation mechanisms in the slope with less runoff volume, we measured the same parameters under the canopy of five *G. scorpius* individuals and in five 3x3m experimental plots (to maintain a similar size of vegetation patches) established in open areas (inter-patches) out of its influence. The experiment lasted for the whole hydrologic year 2007-08, with the installation of the sheets in August 2007, before the arrival of the autumn rains, until September 2008.

Soil moisture measurements were taken periodically (every 15 days without rain and 24 h after each rainfall event) in two points (the upper and the lower part) of each vegetation patch. We monitored the volumetric soil moisture (%) in the first 15 cm of the soil profile following the me thodology proposed by Cassel *et al.* (1994), using a TDR instrument (Tektronix® 1502C), with an accuracy of 94% in the determination of soil moisture. To test the differences in soil water availability between the understorey of *G. scorpius* plants and the open areas in slope 3 we installed TDR sensors horizontally at 5, 25 and 50 cm-depth of the soil profile in each *G. scorpius* vegetation patch and open experimental plot. In this case we used a TDR cable tester to collect the data, following the same methodology described above. Soil water data collection began in November 2007, with the first autumn rains that initiated the hydrologic year.

Leaf water potential of woody species of the vegetation patches was measured in two campaigns: early July and September 2008 (early and late summer respectively), coinciding with the period of maximum water deficit. In slope 1 only *S. chamaecyparissus* was monitored (the unique woody species in the slope) while in slope 3 we obtained data from *G. scorpius*, *S. chamaecyparissus* and *T. vulgaris*. No water potential measurements were taken in slope 2 as no woody species was present in the vegetation patches. Leaf water potential (Ψ, MPa) was determined using a pressure chamber (SKPM 1400, Skye Instruments®), following the methodology proposed by Brown and Tanner (1981). In each campaign, two measurements of water potential were taken at different moments: pre-dawn (from 05:00 to 07:00 a.m.) and mid-day (from 2:00 to 4:00 p.m.). One measure of each species was obtained in each vegetation patch. Additionally, measurements were also taken in slope 3 from individuals of each of the three woody species in each experimental plot out of the *G. scorpius* canopy.

Herb production was measured in late spring by clipping the aerial biomass in five quadrats (10x10cm) randomly distributed in each vegetation patch and open experimental plot. The samples were dried in an oven at 60 ºC for 3 days and then weighed. At the end of the experiment, in September 2008, soil samples (38.5 cm^2 × 4cm depth) were collected to analyze the differences in floristic composition of the soil seed banks of the vegetation patches subjected to different runoff treatments. Four random samples were collected from each vegetation patch of each slope and in each experimental plot out of the *G. scorpius* canopy in slope 3. Each of these samples was subdivided into four subsamples that were placed in 250 ml plastic containers over a 5 cm vermiculite layer. The floristic composition of the soil seed banks was determined by means of germination under optimal conditions in a greenhouse.

Data analysis

Bifactorial repeated measures ANOVAs were used to analyze the differences in soil water content between the vegetation patches along the hydrologic year, with time as within subject factor and runoff treatment and slope as between subject factors in the runoff exclusion experiment, and understorey of *G. scorpius* and soil depth as between subject factors in the facilitation experiment. The same statistical tool was used to explore the differences in leaf water potential of the woody species between treatments, being in this case the two campaigns (July and September) the within subject factor. *Post hoc* Tukey tests were used to determine differences on leaf water potentials between groups. Mann-Whitney U tests were performed to analyze the differences in herb production between treatments. We also calculated the percentage of decrease in biomass produced by the runoff exclusion treatment by means of subtracting the biomass in control vegetation patches from that of their correspondent runoff exclusion patches in each slope. This new variable allowed us to analyze the magnitude of the impact of runoff exclusion between the three slopes, by means of a Kruskall-Wallis test. With data of total density of germinable seeds in the soil seed banks we performed several Mann-Whitney U tests to check differences between treatments. A non-metric multidimensional scaling (NMDS) analysis was performed with data on floristic composition of the soil seed banks. The graphic representation of the samples in the ordination space enabled us to measure the heterogeneity in floristic composition of the vegetation patches subjected to both runoff treatments in the different slopes. We did this by calculating the mean distance (measured through Sorensen index) that each sample maintained with the other samples subjected to the same treatment. A bifactorial ANOVA (with slope and runoff treatment as between subject factors) was performed with this new variable.

All statistics have been carried out with the STATISTICA 6.0 package (Statsoft 2001). Data analyzed through ANOVA and Tukey tests fulfilled parametric assumptions. For the multivariate analysis we used the PC-ORD package (McCune and Mefford 1997). The scientific names of the species are in accordance with *Flora Europaea* (Tutin *et al.* 1964-1980).

Results

The study period was particularly humid, with a total precipitation of 550.67 mm (almost 20% above the historical records), mainly concentrated during the spring (from March to June), when 80% of the total precipitation occurred (Fig. 6.1).

Runoff exclusion experiment

There were differences in soil moisture content in the first 15 cm between the control vegetation patches and those subjected to runoff exclusion (Fig. 6.2), with significant effects of runoff treatment ($F_{1,22}$= 5.07; p=0.03) and time ($F_{19,418}$= 358.58; p<0.01). More water was found in control patches (12.61±0.42, mean ± SE) than in those subjected to runoff exclusion (11.11±0.61, mean ± SE), and also during the spring period. There was also a significant interaction between runoff treatment and slope ($F_{2,22}$= 4.88; p=0.02), as differences between runoff treatments were deeper in slope 1 and decreased in slopes 2 and 3 (Fig. 6.2).

Table 6.2 shows the results of the repeated measures ANOVAs performed with leaf water potential data of the woody species between runoff treatments. In all cases there were significant differences between the two campaigns, being consistently higher the water potentials of July that those of September (Fig. 6.3). Also, water potentials measured at pre-dawn were generally higher than those at midday, although in the case of *S. chamaecyparissus* in slope 1 and in *T. vulgaris* these differences did not reach statistical significance. Runoff exclusion provoked a general reduction in the water potential of all species, especially remarkable in September, at the end of the drought period (Fig. 6.3). However, the statistical significance of this effect was lower in slope 3, in which even *S. chamaecyparissus* did not show these differences. The effect of runoff treatment was always independent of the moment in which measurements were obtained (pre-dawn or midday) and of the campaign (in this latter case only *S. chamaecyparissus* showed a significant interaction with time, as runoff exclusion effects were greater in September campaign).

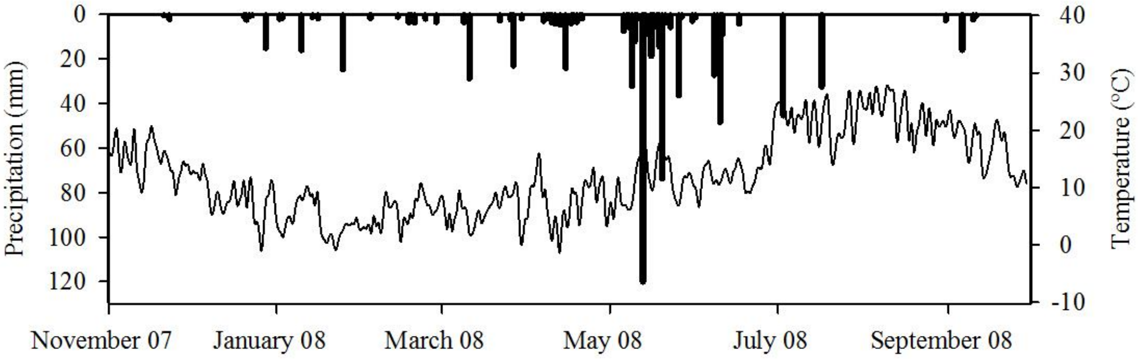


Figure 6.1. Distribution of temperature (line) and precipitation (bars) during the study period.

Table 6.2. Results of the repeated measure ANOVAs performed with leaf water potential data of the woody species in the vegetation patches subjected to both runoff exclusion treatments in the different slopes. Between subject factors are Moment (pre-dawn and midday) and Treatment (control and runoff exclusion). Within subject factor is Time (July and September campaigns). Asterisks indicates significant differences (* at α=0.05 and ** at α=0.01).

	Santolina chamaecyparissus Slope 1		*Thymus vulgaris* Slope 3		*Santolina chamaecyparissus* Slope 3		*Genista scorpius* Slope 3	
	F	**p**	**F**	**p**	**F**	**p**	**F**	**p**
Moment	1.93	n.s.	2.57	n.s.	9.63	**	64.97	**
Treatment	9.65	**	4.45	*	3.17	n.s.	7.54	*
*Moment*Treatment*	0.09	n.s.	0.29	n.s.	0.21	n.s.	0.14	n.s.
Time	118.57	**	75.38	**	229.99	**	143.17	**
*Time*Moment*	13.70	**	1.85	n.s.	39.57	**	1.38	n.s.
*Time*Treatment*	5.29	*	3.58	n.s.	1.73	n.s.	3.49	n.s.
*Time*Moment*Tretament*	0.04	n.s.	1.22	n.s.	0.03	n.s.	0.12	n.s.

Herbaceous biomass was significantly lower in the vegetation patches subjected to runoff exclusion in slope 1 (Fig. 6.4a). Slopes 2 and 3 showed a similar tendency but without statistical significance (Fig. 6.4a). The percentage of biomass decrease between both runoff treatments was significantly different between slopes (Kruskal-Wallis test H=7.98, p=0.02), reaching its highest value in slope 1 and its minimum in slope 3 (Fig. 6.4b). A total of 33 different species were identified in the soil seed banks of the vegetation patches, 10 in slope 1, 14 in slope 2 and 28 in slope 3. No significant differences were found in the density of seeds of the soil seed banks between runoff treatments in none of the slopes. The NMDS performed with the data of floristic composition showed a stress of 12%. The bifactorial ANOVA performed with data of heterogeneity of the soil seed banks (measured through the mean of Sorensen distances between samples from the same treatment in the ordination graph) showed a significant effect of the slope ($F_{2,24}$= 6.93; p<0.01) and runoff treatment ($F_{1,24}$= 4.44; p=0.04). Heterogeneity of soil seed banks was lower in slope 2 (0.45±0.03, mean ± SE) than in slopes 1 (0.61±0.03, mean ± SE) and 3 (0.56±0.03, mean ± SE). Runoff exclusion also provoked a general reduction in the heterogeneity of soil seed banks (0.50±0.03 in runoff exclusion patches *versus* 0.58±0.03 in control ones).

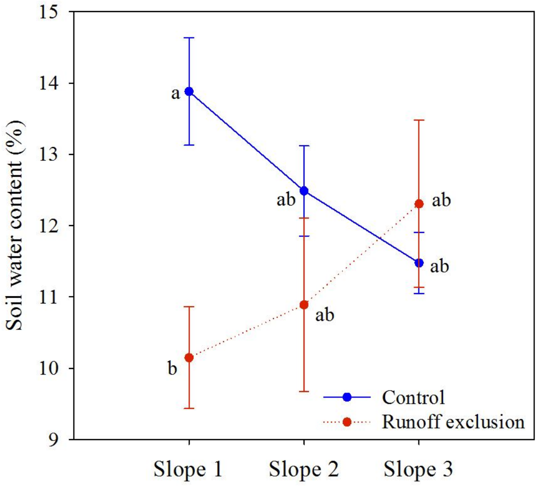


Figure 6.2. Mean soil water content in the first 15 cm in the vegetation patches subjected to runoff treatments in the three slopes. Vertical bars denote 0.95 confidence intervals. Different letters indicate significant differences with the *post hoc* Tukey tests (p<0.05).

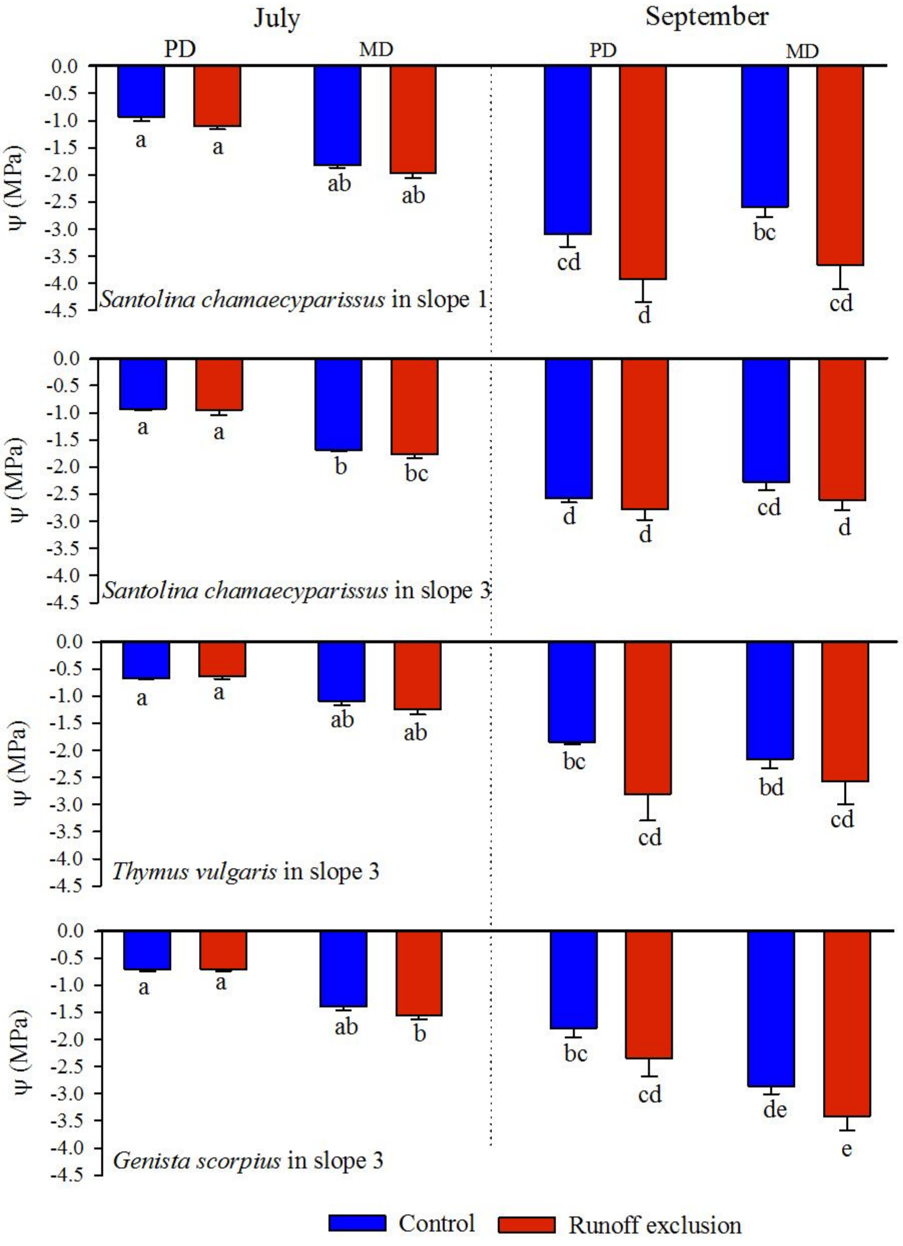


Figure 6.3. Mean leaf water potentials (whiskers indicate the standard error) of the woody species of the vegetation patches in the campaigns of July and September. PD: pre-dawn, MD: midday. Different letters indicate significant differences with the *post hoc* Tukey tests ($p<0.05$).

Facilitation mechanism in slope 3

The repeated measures ANOVA with the data of soil moisture in the vegetation patches under *G. scorpius* canopy and in open areas of slope 3 showed significant effects of sensor depth in the soil profile ($F_{2,17}$= 16.87; p<0.01) and time ($F_{19,323}$= 370.96; p<0.01). Soil water content was higher at 50 cm depth (27.39±0.51, mean ± SE) than at 25 cm (24.14±1.19, mean ± SE) or 5 cm (18.48±1.20, mean ± SE). As regards time, soil water content was always higher during the spring period (Fig. 6.5). There was a significant interaction between time and sensor depth ($F_{38,323}$=10.69; p<0.01) as differences in soil water content along the soil profile become more patent from middle spring onwards (Fig. 6.5). No significant effect of the *G. scorpius* understorey was found ($F_{1,17}$= 0.77; p=0.39).

Results of the repeated measures ANOVAs performed with leaf water potential data of the woody species of slope 3 under *G. scorpius* canopy and in open areas are shown in Table 6.3. As in the case of the runoff exclusion experiment, all water potentials were highly influenced by the time in which each campaign was performed, with water potentials of September clearly lower than those of July (Fig. 6.6). Location under the canopy of *G. scorpius* individuals increased significantly the water potentials of *S. chamaecyparissus* and *T. vulgaris*, in the latter case this effect was also dependent on time, as greater differences were found in September (Fig. 6.6).

Herb production under the canopy of *G. scorpius* individuals was significantly higher than that in open areas (Mann-Whitney U=3.00, p=0.05), with values of 281.97±34.9 g cm^{-2} (mean ± SE) under *G. scorpius* and 177.87±23.9 g cm^{-2} (mean ± SE) in open areas. Seed density of the soil seed banks was also higher in the understorey of *G. scorpius* (7844.1±3061.8 seeds m-2, mean ± SE) than in open areas (1493.5±493.2 seeds m^{-2}, mean ± SE) (Mann-Whitney U=2.00, p=0.03).

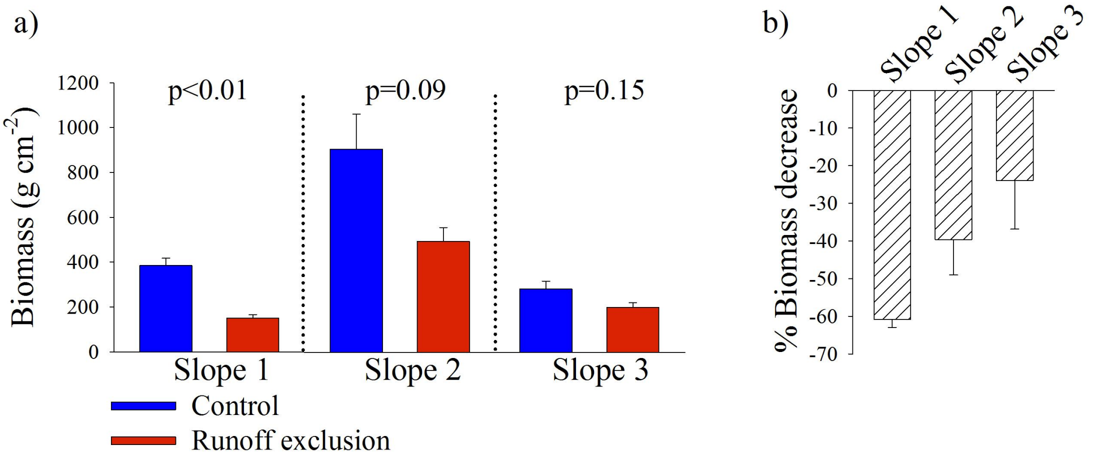


Figure 6.4. a) Mean herbaceous production (whiskers indicate the standard error) in the vegetation patches subjected to different runoff treatments in the three slopes. p values of Mann-Whitney U tests are indicated. b) Mean percentage of biomass decrease between control and runoff exclusion treatments in each slope.

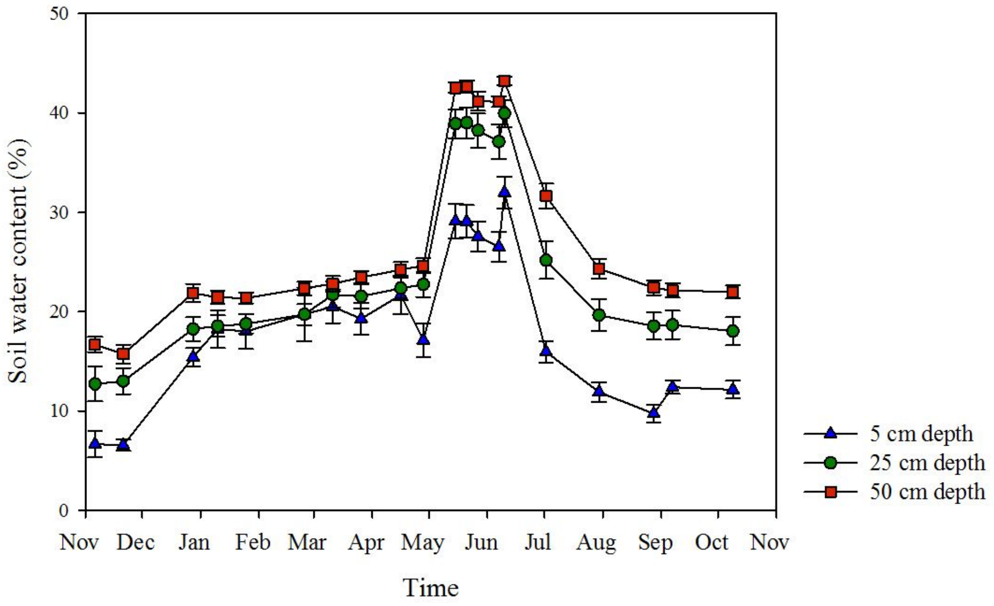


Figure 6.5. Mean soil water content in the vegetation patches of slope 3 (understorey of *Genista scorpius* and open areas averaged) at 5, 25 and 50cm depth during the study period.

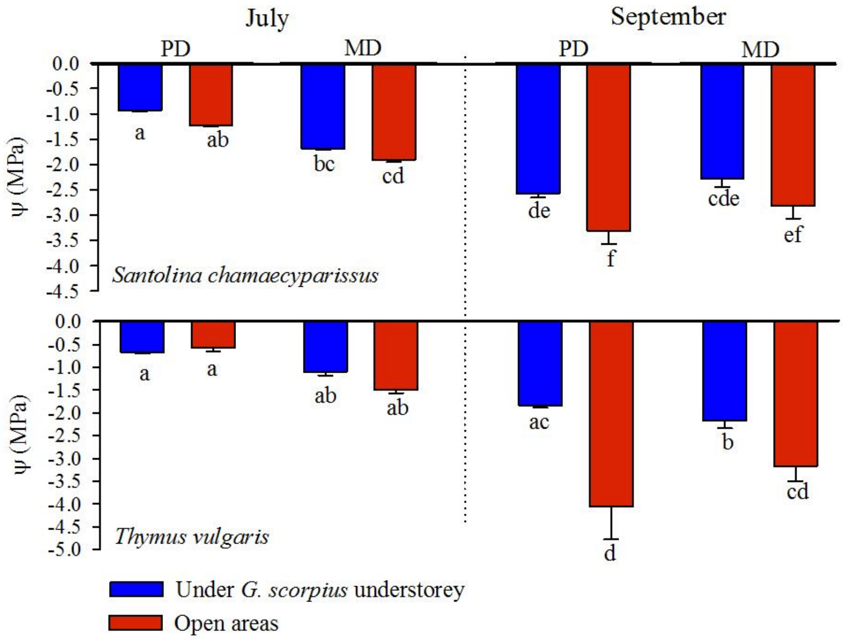


Figure 6.6. Mean leaf water potentials (whiskers indicate the standard error) of the woody species of slope 3 in the campaigns of July and September. PD: pre-dawn, MD: midday. Different letters indicate significant differences with the *post hoc* Tukey tests (p<0.05).

Discussion

Experimental manipulation of runoff flow between runoff source and sink patches in artificial slopes subjected to common water limitations of the Mediterranean area allowed us to demonstrate the importance of such ecohydrological interactions for the dynamics of these ecosystems. In general, runoff coming from inter-patches upslope improved the hydric status of the vegetation in the patches that received this water as runon, as higher leaf water potentials were found in control patches than in those where runoff was excluded. Similar results were found by Seghieri and Galle (1999) in a runoff exclusion experiment in the banded vegetation of Niger: leaf water potentials of the dominant shrubs *Guiera senegalensis* and *Combretum micranthum* decreased in the plots subjected to runoff exclusion. Our results are more significant if we consider that this hydrologic year was extraordinarily wet, especially during spring, as leaf water potentials are highly influenced by seasonality (factor time in our statistical analysis). Although evidence for the importance of runoff redistribution between vegetation patches in the dynamics of semiarid ecosystems has been indicated by other authors (Ludwig *et al*. 2005, Wilcox *et al*. 2003), this is the first time in which empirical data on the effect of these interactions during the recovery process in artificial ecosystems is reported.

Significant differences in soil water content (Fig. 6.2) and in herb production (Fig. 6.4) were only found in slope 1, where runoff exclusion seriously affected both parameters. Others authors have found similar results in natural ecosystems, for example, Noble *et al.* (1998) studied natural mulga woodlands of Australia and observed a depress in *Thyridolepis mitchelliana* (mulga grass) dry matter when the access of overland flow was prevented. Also, Schlesinger and Jones (1984) found less biomass and density of the shrubs *Larrea tridentata* and *Ambrosia dumosa* in areas of the Mojave desert that where isolated from overland flow by the ditches erected in 1936 during the construction of the Colorado river aqueduct. The runoff gradient incorporated in our experimental design permitted us to demonstrate that ecological effects of runoff redistribution are highly dependent on the total runoff volume routed by the slopes. In fact, we found that the effects of runoff exclusion were

Table 6.3. Results of the repeated measure ANOVAs performed with leaf water potential data of the woody species under different facilitation situations in slope 3. Between subject factors are Moment (pre-dawn and midday) and Treatment (*Genista* understorey or open areas). Within subject factor is Time (July and September campaigns). Asterisks indicates significant differences (* at α=0.05 and ** at α=0.01).

	Thymus vulgaris		*Santolina chamaecyparissus*	
	F	**p**	**F**	**p**
Moment	0.94	n.s.	2.23	n.s.
Treatment	18.46	**	16.10	**
*Moment*Treatment*	0.75	n.s.	0.41	n.s.
Time	80.08	**	199.30	**
*Time*Moment*	5.26	*	35.72	**
*Time*Treatment*	12.31	**	4.00	n.s.
*Time*Moment*Treatment*	4.37	n.s.	0.11	n.s.

more intense in slope 1 (where total runoff coefficient was 11.17%) and lower in slope 2 and 3, with runoff coefficients of 2.04 and 0.5%, respectively (Table 6.1). Changes in the percentage of decrease in herb production produced by runoff exclusion between the three slopes clearly reflect this tendency (Fig. 6.4b). The same conclusion derives also from the fact that runoff exclusion provoked a large decrease in the leaf water potential of *S. chamaecyparissus* in slope 1 but not in slope 3 (Table 6.2, Fig. 6.3), being the only woody species that appeared in both slopes.

As total runoff decreases, the runoff-runon interactions between patches become less intense at the same time that positive interactions between plants became the driving force in structuring the vegetation, as patches of *G. scorpius* enhance performance of vegetation under its canopy in the slope 3. In fact, 63% more herb production was found in the undestorey of *G. scorpius* shrubs than in open areas. Also, leaf water potentials of *T. vulgaris* and *S. chamaecyparissus* and seed density of the soil seed banks were higher under *G. scorpius* individuals. The fact that we did not find differences in soil moisture related to *G. scorpius* canopies must be interpreted as a result of water transpiration through the herbaceous community, especially if we consider that soil water availability shows its lowest values at shallow depths (Fig. 6.5).

Thus, *G. scorpius* patches would act as fertility islands (Garner and Steinberger, 1989) in the constructed slopes, or, in more ecohydrological terms, as "islands of hydrologically enhanced biotic productivity", according to Rango *et al.* (2006). In fact, differences in edaphic properties that improve water infiltration such as bulk density and soil surface strength has been found under *G. scorpius* canopies (Table 6.1). Results from other authors also point to the idea that facilitation in semiarid environments is mainly mediated by positive effects on water availability (Maestre *et al.* 2001).

The appearance of facilitation processes in the slope subjected to a lower volume of runoff indicates the predominance of biotic factors over abiotic ones. This can be interpreted as a signal of ecosystem recovery, as usually the shift from biotic to abiotic control of water takes place as ecosystems become degraded (Davenport *et al.* 1998; Schlesinger *et al.* 1990). Besides, facilitation is widely recognized as a driving force in semiarid environments (Kéfi *et al.* 2007) as it is considered one of the main mechanisms that allow the self-organization of these ecosystems (Solé 2007).

The fact that we observe facilitation processes in the least degraded slope (with the minimum volume of runoff) seems to contradict traditional models of plant-plant interaction that predict increasing facilitation intensity with stress increases (Bertness and Callaway 1994). Our results agree with those of Maestre *et al.* (2005) who stated that in water limited environments facilitation can only occur when facilitators increase water availability beyond their own uptake requirements. It can be concluded that the limitations imposed by water stress in these artificial slopes are so high that only large shrubs such as *G. scorpius* can modify the conditions under their understorey in order to facilitate the establishment of other plants.

Therefore, *G. scorpius* can be considered a key species for the ecological succession in these reclaimed landscapes in a semiarid environment. We have found patches of this species only in the slope subjected to the minimum runoff volume, thus following a similar trend of natural succession on abandoned agricultural lands in the area, where the dominance of *G. scorpius* occurs between 10 and 35 years since abandonment (Molinillo *et al.* 1997). High amounts of runoff hinder the recovery of constructed slopes by triggering soil erosion processes, as we have observed in other slopes of the same area (Espigares *et al.* in press; Moreno-de las Heras *et al.* 2010). *G. scorpius* contributes to the advance of the ecological succession in the reclaimed slopes by favoring the biotic control of water in the system. Similarly, Maestre and Cortina (2004) have identified other shrubs as keystone species because of their role in infiltration and nutrient cycling in semiarid ecosystems of SE Spain. Therefore, it would be very convenient to introduce these species in early phases of restoration projects and avoid the mere revegetation with fast growing species that could afterwards arrest the process of ecological succession (Moreno-de las Heras *et al.* 2008).

The analysis of the ecohydrological interactions between vegetation patches in artificial slopes in a runoff gradient offers important implications for the design of restoration projects in semiarid environments. Some authors have stated that the best way to rehabilitate dysfunctional semiarid ecosystems is to restore vegetated patches like structures that best trap and store limited soil resources (Ludwig *et al.* 1999). One of the few experiences following this recommendation has been the establishment of runoff catcher structures in degraded *Acacia aneura* forest in Australia, by means of piles of large tree branches and shrubs located along contour lines. This improved soil properties (Tongway and Ludwig 1996) and increased perennial species abundance (Ludwig and Tongway 1996). Also, Manu *et al.* (2000) described a successful project of restoration of degraded tiger bush vegetation in Niger with microcatchments runoff harvesters and the introduction of selected woody species. However, more investigation on the ecohydrological interactions during the recovery process in reclaimed ecosystems is needed to improve the efficiency of restoration practices in drylands.

Conclusions

We have demonstrated that ecohydrological interactions between runoff source and sink patches is crucial for the dynamics of artificial slopes in water limited environments, as improvements in the water status of soils and vegetation have been found. Also, this effect is more intense with runoff increases, provided that thresholds that initiate intense soil erosion processes have not been surpassed. We have also observed that biotic control of water increases as total runoff volume decreases because vegetation patches of large shrubs (*G. scorpius* in our Mediterranean environment) create islands of hydrologically enhanced biotic productivity. Some important implications for dryland restoration derive from these results, as high amounts of runoff, that usually characterize the early phases of these artificial ecosystems, could be redirected to target nuclei in which introduced key species would benefit from this extra water supply. Through their capacity to improve soil properties and to enhance vegetation establishment, these shrub patches would accelerate the ecological succession while preventing the loss of water and soil resources produced by soil erosion. The concept of "runoff expert management" emerges as an important issue for restoration, with implications for the design of topography and revegetation of reclaimed ecosystems.

Acknowledgments

This work was supported by the Universidad de Alcalá and the project REMEDINAL (S2009AMB-1783), funded by the Regional Government of Madrid. We are also grateful to Jose Antonio Merino-Sánchez for his laboratory help and to Jesús Romero-Trillo for language editing.

References

Aguiar MR, Sala OE. 1999. Patch structure, dynamics and implications for the functioning of arid ecosystems. Trends in Ecology & Evolution 14: 273-277.

Bautista S, Mayor AG, Bourakhouadar J, Bellot J. 2007. Plant spatial pattern predicts hillslope runoff and erosion in a semiarid Mediterranean landscape. Ecosystems 10: 987-998.

Bertness MD, Callaway RM. 1994. Positive interactions in communities. Trends in Ecology & Evolution 9: 191-193.

Bhark EW, Small EE. 2003. Association between plant canopies and the spatial patterns of infiltration in shrubland and grassland of the Chihuahuan desert, New Mexico. Ecosystems 6: 185-196.

Brown PW, Tanner CB. 1981. Alfalfa water potential measurement: a comparison of the pressure chamber and leaf dew-point hygrometers. Crop Science 21: 240-244.

Calvo-Cases A, Boix-Fayos C, Imeson AC. 2003. Runoff generation, sediment movement and soil water behaviour on calcareous (limestone) slopes of some Mediterranean environments in southeast Spain. Geomorphology 50: 269-291.

Cammeraat LH, Imeson AC. 1999. The evolution and significance of soil-vegetation patterns following land abandonment and fire in Spain. Catena 37: 107-127.

Cassel DK, Kachanoski RG, Topp GC. 1994. Practical considerations for using a TDR cable tester. Soil Technology 7: 113-126.

Cornet AF, Montaña C, Delhoume JP, Lopez-Portillo J. 1992. Water flows and the dynamics of desert vegetation stripes. Hansen AJ, Di Castri F, editors. Landscape boundaries. Consequences for biotic diversity and ecological flows. Berlin: Springer-Verlag. p 327-345.

Davenport DW, Breshears DD, Wilcox BP, Allen CD. 1998. Viewpoint: Sustainability of pinon-juniper ecosystems - a unifying perspective of soil erosion thresholds. Journal of Range Management 51: 231-240.

Dunkerley DL, Brown KJ. 1995. Runoff and runon areas in a patterned chenopod shrubland, arid western New South Wales, Australia: characteristics and origin. Journal of Arid Environments 30: 41-55.

Espigares T, Moreno-de las Heras M. Nicolau JM. Performance of vegetation in reclaimed slopes affected by soil erosion. Restoration Ecology, in press. doi: 10.1111/j.1526-100X.2009.00546.x.

Garner W, Steinberger Y. 1989. A proposed mechanism for the formation of 'Fertile Islands' in the desert ecosystem. Journal of Arid Environments 16: 257-262.

Hanna DM, Sadler JP, Wood PJ. 2007. Hydroecology and ecohydrology: a potential route forward? Hydrological Processes 21: 3385-3390.

Hobbs RJ, Arico S, Aronson J, Baron JS, Bridgewater P, Cramer VA, Epstein PR, Ewel JJ, Klink CA, Lugo AE, Norton D, Ojima D, Richardson DM, Sanderson EW, Valladares F, Vilà M, Zamora R, Zobel M. 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15: 1-7.

Hobbs RJ, Higgs E, Harris JA. 2009. Novel ecosystems: implications for conservation and restoration. Trends in Ecology and Evolution 24: 599-605.

Imeson AC, Prinsen HAM. 2004. Vegetation patterns as biological indicators for identifying runoff and sediment source and sink areas for semi-arid landscapes in Spain. Agriculture Ecosystems & Environment 14: 333-342.

Kéfi S, Rietkerk M, Alados C, Pueyo Y, Papanastasis V, ElAich A, de Ruiter PC. 2007. Spatial vegetation patterns and imminent desertification in Mediterranean arid ecosystems. Nature 449: 213-217.

Lavee H, Imeson AC, Sarah P. 1998. The impact of climate change on geomorphology and desertification along a Mediterranean-arid transect. Land Degradation & Development 9: 407-422.

Ludwig JA, Tongway D. 1995. Spatial organisation of landscapes and its function in semi-arid woodlands, Australia. Landscape Ecology 10: 51-63.

Ludwig JA, Tongway D. 1996. Rehabilitation of semiarid landscapes in Australia. II. Restoring vegetation patches. Restoration Ecology 4: 398-406.

Ludwig JA, Tongway D, Marsden SG. 1999. Stripes, strands or stipples: modelling the influence of three landscape banding patterns on resource capture and productivity in semiarid woodlands, Australia. Catena 37: 257-273.

Ludwig JA, Wilcox BP, Breshears DD, Tongway DJ, Imeson AC. 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology 86: 288-297.

Maestre FT, Bautista S, Cortina J, Bellot J. 2001. Potential for using facilitation by grasses to establish shrubs on a semiarid degraded steppe. Ecological Applications 11: 1641-1655.

Maestre FT, Cortina J. 2004. Insights into ecosystem composition and function in a sequence of degraded semiarid steppes. Restoration Ecology 12: 494-502.

Maestre FT, Valladares F, Reynolds JF. 2005. Is the change of plant-plant interactions with abiotic stress predictable? A meta-analysis of field results in arid environments. Journal of Ecology 93: 748-757.

Manu A. Thurow TL, Juo A, Zanguina I. 2000. Agroecological impacts of a practical approach for restoration of degraded Sahelian watersheds. Lal R, editor. Integrated Watershed Management in the Global Ecosystem. Florida: CRC Press. p 145-163.

McCune B, Mefford MJ. 1997. Multivariate analysis of ecological data version 3.18. Gleneden Beach (OR): MJM Software. xxiii+197p.

Molinillo M, Lasanta T, García-Ruiz JM. 1997. Managing mountainous degraded landscapes after farmland abandonment in the central Spanish Pyrenees. Environmental Management 21: 587-598.

Moreno-de las Heras M. 2009. Development of soil physical structure and biological functionality in mining spoils affected by soil erosion in a Mediterranean-continental environment. Geoderma 149: 249-256.

Moreno-de las Heras M, Merino-Martín L, Nicolau JM. 2009. Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. Catena 77: 39-47.

Moreno-de las Heras M, Nicolau JM, Espigares T. 2008. Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecological Engineering 34: 168-178.

Moreno-de las Heras M, Nicolau JM, Merino-Martín L, Wilcox BP. 2010. Plot-scale effects on runoff and erosion along a slope degradation gradient. Water Resources Research 46: W04503. doi:10.1029/2009WR007875.

Newman BD, Wilcox BP, Archer SR, Breshears DD, Dahm CN, Duffy CJ, McDowell NG, Phillips FM, Scanlon BR, Vivoni ER. 2006. Ecohydrology of water-limited environments: A scientific vision. Water Resources Research 42: doi:10.1029/2005WR004141.

Nicolau JM, Asensio E. 2000. Rainfall erosion on opencast coal-mine lands. An ecological perspective. Land Reconstruction and Management 1: 51-73.

Noble JC, Greene RSB, Müller WJ. 1998. Herbage production following rainfall redistribution in a semi-arid mulga (*Acacia aneura*) woodland in western New South Wales. Rangeland Journal 20: 206-225.

Noy-Meir I. 1973. Desert ecosystems: environment and producers. Annual Review of Ecology and Systematics 4: 25-51.

Papadakis J. 1966. Climates of the world and their agricultural potentialities. Buenos Aires. 174p.

Porporato A, Rodríguez-Iturbe I. 2002. Ecohydrology-a challenging multidisciplinary research perspective. Hydrological Sciences Journal 47: 811-821.

Puigdefábregas J, Sole A, Gutiérrez L, Barrio G, Boer M. 1999. Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in Southeast Spain. Earth-Science Reviews 48: 39-70.

Rango A, Tartowskia SL, Lalibertea A, Wainwright J, Parsons A. 2006. Islands of hydrologically enhanced biotic productivity in natural and managed arid ecosystems. Journal of Arid Environments 65: 235-252.

Reid KD, Wilcox BP, Breshears DD, MacDonald L. 1999. Runoff and erosion in a Piñon-Juniper woodland: influence of vegetation patches. Soil Science Society of America Journal 63: 1869-1879.

Scanlon TM, Caylor KK, Levin SA, Rodriguez-Iturbe I. 2007. Positive feedbacks promote power-law clustering of Kalahari vegetation. Nature 449: 209-211.

Schlesinger WH, Jones CS. 1984. The comparative importance of overland runoff and mean annual rainfall to shrub communities of the Mojave desert. Botanical Gazetta 145: 116-124.

Schlesinger WH, Reynolds JF, Cunningham GL, Huenneke LF, Jarrell WM, Virginia RA, Whitford WG. 1990. Biological feedbacks in global desertification. Science 247: 1043-1048.

Seghieri J, Galle S. 1999. Run-on contribution to a Sahelian two-phase mosaic system: soil water regime and vegetation life cycles. Acta Oecologica 20: 209-217.

Seghieri J. Galle S, Rajot JL, Ehrmann M. 1997. Relationships between soil moisture and growth of herbaceous plants in a natural vegetation mosaic in Niger. Journal of Arid Environments 36: 87-102.

StatSoft Inc. 2001. Statistica for Windows (Computer Program Manual). Tulsa (OK): StatSoft Inc.

Solé R. 2007. Scaling laws in the drier. Nature 449: 151-153.

Tongway D, Ludwig JA. 1996. Rehabilitation of semiarid landscapes in Australia. I. Restoring Productive soils. Restoration Ecology 4: 388-397.

Turnbull L, Wainwright J, Brazier RE. 2008. A conceptual framework for understanding semi-arid land degradation: ecohydrological interactions across multiple-space and time scales. Ecohydrology 1: 23-34.

Tutin TG, Heywood VH, Walters SM, Webb DA. 1964-1980. Flora Europaea. Cambridge (UK): Cambridge University Press. Vols 1-5.

van de Koppell J, Rietkerk M. 2004. Spatial interactions and resilience in arid ecosystems. American Naturalist 163: 113-121.

Wilcox BP, Breshears DD, Allen CD. 2003. Ecohydrology of a resource-conserving semiarid woodland: effects of scale and disturbance. Ecological Monographs 73: 223-239.

Wilcox BP, Thurow TL. 2006. Emerging issues in rangeland ecohydrology: vegetation change and the water cycle. Rangeland Ecology & Management 59: 220-224.

White LP. 1969. Vegetation arcs in Jordan. Journal of Ecology 57: 461-464.

White LP. 1970. Brousse tigre'e patterns in southern Niger. Journal of Ecology 58: 549-553.

Capítulo 7

A Vir

And sometimes I get nervous
When I see an open door
Close your eyes, clear your heart
Cut the cord

Are we human or are we dancer?
My sign is vital, my hands are cold
And I'm on my knees looking for the answer
Are we human or are we dancer?

Human, The Killers

Capítulo 7

Síntesis general

En este capítulo se hace una síntesis general de los principales resultados obtenidos en los trabajos desarrollados en los capítulos 3 a 6 de la Tesis Doctoral.

En la presente Tesis Doctoral se han estudiado las principales interacciones entre hidrología y vegetación en laderas restauradas de la minería de carbón a cielo abierto. Estas laderas se encuentran en proceso de sucesión ecológica y presentan diferente volumen de escorrentía circulante a causa de su diseño topográfico.

La hipótesis general de partida es que la interacción entre la escorrentía superficial y la vegetación es un fenómeno clave para explicar la estructura espacial y el funcionamiento de estos sistemas y, en particular, que la escorrentía superficial constituye un factor director de la sucesión de laderas restauradas (agradación). Con los diferentes trabajos científicos ejecutados en esta Tesis se ha testado esta hipótesis junto con otras hipótesis secundarias desarrolladas en el capítulo 1.

En trabajos anteriores realizados en laderas restauradas de minería de carbón a cielo abierto ya se habían descrito las relaciones ecohidrológicas de laderas con intensos procesos de reguerización (Nicolau 2002; Moreno-de las Heras *et al*. 2008; Moreno-de las Heras 2009a, b; Moreno-de las Heras *et al*. 2009; Moreno-de las Heras *et al*. 2010), en las que la sucesión se ve muy limitada (*arrested succession*). Sin embargo, son muy escasos los trabajos que analizan la ecohidrología de laderas restauradas en las que la escorrentía circula de forma laminar, pudiendo jugar un papel favorecedor de la sucesión vegetal.

En una primera aproximación (capítulo 3), se ha demostrado que en estas laderas la escorrentía y la vegetación presentan una intensa interrelación. Esta interacción hidrología-vegetación configura a escala de mancha unas unidades que hemos denominado "unidades ecohidrológicas" que se caracterizan por una composición florística y un comportamiento hidrológico determinados que son estudiados en los capítulos posteriores. Asimismo, como ya se ha determinado en trabajos anteriores (Moreno-de las Heras 2009b), se ha identificado la reguerización como un proceso de degradación en el que el control del agua recae sobre los procesos abióticos. También se ha puesto de manifiesto que la intensidad de las interrelaciones ecohidrológicas es menor en las laderas cubiertas de vegetación, puesto que no hay procesos de escorrentía superficial al estar los procesos hidrológicos fundamentalmente controlados por la vegetación.

Posteriormente se ha abordado la investigación a escala de ladera con el objetivo de evaluar los efectos del volumen de escorrentía en las interacciones ecohidrológicas durante el proceso de sucesión (agradación) de estas laderas restauradas. Para ello se seleccionaron laderas con similares tratamientos de restauración aplicados (sustrato, revegetación, orientación) que diferían en la topografía de cabecera, la cual determina la superficie generadora de escorrentía. De esta forma, se seleccionaron tres laderas a lo largo de un gradiente de perturbación por escorrentía, desde una situación de reguerización discontinua hasta una escorrentía laminar débil. Esta situación propiciada por las prácticas de restauración realizadas en la mina de "El Moral" se puede considerar como una de las más favorables que pueden encontrarse en la naturaleza para estudiar de manera aislada el efecto de la escorrentía sobre los procesos ecohidrológicos, sin llegar a constituir, ciertamente, un experimento con réplicas auténticas.

En los capítulos 4, 5 y 6 se ha utilizado el marco conceptual desarrollado por Ludwig (1997; 2005) para ambientes semiáridos naturales a fin de analizar las interacciones ecohidrológicas en las laderas restauradas. Este marco conceptual ha sido ampliamente comprobado para ambientes naturales, tanto en patrones en bandas (Dunkerley y Brown 1999; Galle *et al*. 1999; Ludwig *et al*. 1999; Seghieri y Galle 1999; Valentin *et al*. 1999), como en patrones en mosaico (Cerdà 1997; Puigdefábregas *et al*. 1999; Puigdefábregas 2005). Asimismo, se han estudiado los efectos de las perturbaciones sobre las interacciones descritas por este marco conceptual (McIvor *et al*. 1995; Scanlan *et al*. 1996; Wilcox *et al*. 2003). Sin embargo, este modelo aún no había sido aplicado a ambientes restaurados, donde puede tener una utilidad práctica para el desarrollo de estrategias de restauración adecuadas. De las interacciones tenidas en cuenta por Ludwig y colaboradores, en esta Tesis se han analizado las cuatro identificadas como fundamentales por Ludwig *et al*. (2005):

1.- Las manchas de vegetación obstruyen la escorrentía y almacenan agua.

Esta interacción se ha descrito en los capítulos 4 y 5. En estos capítulos se ha comprobado la existencia de menores tasas de escorrentía y un mayor contenido de humedad en las manchas que actúan como sumideros de escorrentía en comparación con las que se comportan como fuentes.

Además, se ha identificado una tipología muy variada de sumideros y fuentes (capítulo 4). En el caso de los sumideros, se ha descrito un sumidero que destaca por la menor generación de escorrentía y sedimentos, un mayor contenido de humedad y una mayor capacidad de infiltración que hemos denominado "sumidero profundo" por su capacidad de introducir el agua en profundidad. Este sumidero está asociado a la presencia de la especie *Genista scorpius*, un matorral con una diversa cohorte de plantas bajo su dosel. El otro tipo de sumideros son aquéllos en los que las condiciones de infiltración y de generación de escorrentía y sedimentos no son tan patentes (denominados "sumideros en derramaderos de escorrentía"). Asimismo, se ha observado una relación entre los tipos de sumideros descritos y los procesos de formación de patrones de vegetación en ambientes semiáridos (capítulo 4). Los procesos de formación de patrones son dos fundamentalmente: patrones dirigidos por la geomorfología (p. ej. Wainwright *et al*. 2002) o patrones dirigidos por la acción de la vegetación (p. ej. Puigdefábregas 2005).

1a.- Patrones dirigidos por la geomorfología: Cuando los volúmenes de escorrentía circulante a escala de ladera superan un determinado umbral como para generar geoformas consecuencia de los procesos de erosión-transporte-sedimentación (conos de deyección de regueros y derramaderos de escorrentía), los sumideros que se generan son los "sumideros en derramaderos de escorrentía". Dentro de éstos, se han identificado dos tipos en las laderas en estudio:

En la ladera 1, con regueros discontinuos, las geoformas que se han formado son conos de deyección de regueros, en las que se desarrollan unos sumideros que obstruyen la escorrentía y los sedimentos de forma muy limitada. En estos conos subsisten manchas de *Lolium perenne* que es una especie remanente de la siembra inicial que permanece en estas zonas en las que existe un aporte extra de agua (capítulos 4 y 5).

En la ladera 2, con un flujo de escorrentía laminar medio sin desarrollo de regueros, se han formado derramaderos de escorrentía en las zonas de rotura de circulación preferencial de escorrentía que se caracterizan por la presencia de *Brachypodium retusum*, que ha colonizado estos inestables micro-ambientes sedimentarios y de mayor disponibilidad de agua (capítulos 4 y 5).

1b.- Patrones dirigidos por la vegetación: Cuando los volúmenes de escorrentía no han generado microgeoformas (es el caso de la ladera 3), se ha observado el desarrollo de un "sumidero profundo" constituído por matorrales de *Genista scorpius*. Este sumidero, que inicialmente se distribuye al azar posteriormente produce modificaciones en el sustrato bajo su dosel, mejorando las condiciones de infiltración y generando un microambiente favorable para el establecimiento de las plantas (capítulo 5).

2.- El agua de escorrentía provoca un pulso en el crecimiento de la vegetación.

Esta interacción ha sido comprobada en los capítulos 5 y 6. En el capítulo 5 se ha mostrado que unos mayores contenidos de humedad se relacionan con una mayor riqueza florística de la comunidad y del banco de semillas. En el capítulo 6 el experimento de exclusión de escorrentía ha permitido evaluar los efectos del agua extra que llega a las manchas sumidero por la redistribución de la escorrentía sobre los contenidos de humedad, la producción de biomasa y el estrés de las plantas, observando cómo existe una dependencia funcional entre los sumideros y el agua generada en los claros situados por encima. En conclusión, se ha observado que el aporte de escorrentía desde las áreas-fuente influye positivamente en los sumideros, que ven así incrementado su rendimiento tanto a escala de comunidad (riqueza y composición florística), como a escala de planta (producción y estrés hídrico).

3.- Las manchas de vegetación mejoran la capacidad de infiltración del suelo.

Esta interacción se ha comprobado en el capítulo 5. En este capítulo se han realizado simulaciones de lluvia caracterizando las tasas de infiltración finales y los coeficientes de escorrentía a escala de mancha. Asimismo, se han realizado medidas de las propiedades fisicoquímicas del suelo, en las que se ha comprobado que los sumideros, especialmente el sumidero profundo de *Genista scorpius*, tienen unas propiedades físicas (densidad aparente y dureza de costra) más apropiadas para una infiltración eficiente, junto con unas propiedades químicas de fertilidad (N, P, m.o.) que favorecen la formación de agregados, lo que mejora de forma significativa las condiciones de infiltración.

4.- Las perturbaciones influyen en las salidas de escorrentía a escala de ladera.

Tradicionalmente, el estudio de los efectos de las perturbaciones en el marco conceptual TTRP se ha abordado describiendo cómo las perturbaciones (fundamentalmente pastoreo y fuego) afectan negativamente a los sumideros de escorrentía y con ello a las salidas de agua a escala de ladera (McIvor *et al*. 1995; Scanlan *et al*. 1996; Wilcox *et al*. 2003). Sin embargo, en el caso de esta Tesis Doctoral se ha procedido de forma inversa: se ha estudiado cómo afectan los procesos que suceden a escala de ladera sobre los procesos que se desarrollan a escala de mancha. De esta manera, los capítulos 4, 5 y 6 de la Tesis Doctoral están contextualizados en un gradiente de escorrentía, lo que permite analizar los efectos del volumen de escorrentía que circula por la ladera sobre las interacciones ecohidrológicas a escala de mancha.

De forma general, se ha identificado la escorrentía como un factor director de la sucesión ecológica en laderas restauradas de ambientes semiáridos. En el capítulo 4, la escorrentía se ha identificado como un factor que influye en lo que hemos denominado "diversidad hidrológica". Esta "diversidad hidrológica" se refiere a una diversidad de tipos de comportamientos hidrológicos (desde fuentes a sumideros). Un mayor volumen de escorrentía influye en la diversidad de tipos hidrológicos, no sólo disminuyendo su diversidad, sino provocando una dicotomía extrema de los comportamientos (fuentes muy extremas o sumideros). En situaciones de volúmenes reducidos de escorrentía no sólo se observa una mayor "diversidad hidrológica", sino que los comportamientos son más atemperados y además, en el caso de la ladera de menor volumen de escorrentía, aparece un "sumidero profundo" (lo que representa un importante cambio cualitativo).

En el capítulo 5 se demuestra que el volumen de escorrentía circulante por la ladera está asociado con las probabilidades de colonización y de establecimiento de la vegetación. Unos mayores volúmenes de escorrentía incrementan el déficit hídrico (puesto que generan geoformas que evacúan la escorrentía), disminuyendo no sólo las probabilidades de germinación, sino también las probabilidades de establecimiento de las plantas por su influencia en la humedad profunda. Este hecho se ha comprobado a dos escalas, a escala de ladera, viendo cómo se reduce la disponibilidad de agua para las plantas; y a escala de mancha, viendo cómo afecta el volumen de escorrentía a la humedad de las manchas y así, a las probabilidades de germinación y al establecimiento de la vegetación.

En el capítulo 6 ha quedado establecido que un mayor volumen de escorrentía circulante a escala de ladera influye en la funcionalidad de la redistribución de escorrentía. En laderas donde el volumen de escorrentía circulante es mayor, los sumideros de escorrentía son más dependientes de la escorrentía procedente de los claros. Además, se ha identificado la aparición de procesos de facilitación en la ladera con volúmenes de escorrentía más bajos (ladera 3).

Los resultados obtenidos por esta Tesis Doctoral sugieren que a mayores volúmenes de escorrentía (laderas 1 y 2) las manchas de vegetación son más dependientes de la escorrentía y el control del agua se ejerce de forma abiótica. Por otro lado, a volúmenes bajos de escorrentía, el control del agua recae sobre fenómenos de tipo biótico, apareciendo un matorral (*Genista scorpius*) capaz de modificar el patrón de redistribución de la escorrentía y que da lugar a interacciones bióticas (facilitación) que pueden propiciar un total cubrimiento de la ladera por parte de la vegetación (Figura 7.1).

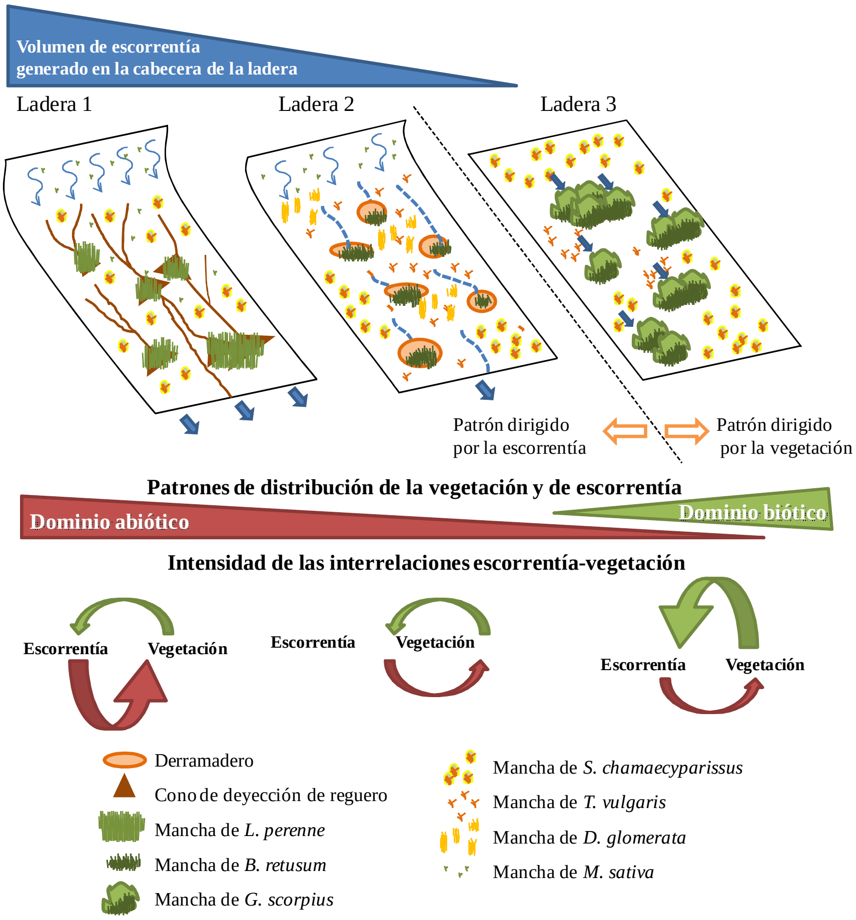


Figura 7.1. Diagrama que representa el efecto de la escorrentía generada en la cabecera sobre las interrelaciones ecohidrológicas de las laderas del estudio.

Referencias bibliográficas

Cerdà A. (1997). The effect of patchy distribution of Stipa tenacissima L on runoff and erosion. Journal of Arid Environments, 36, 37-51.

Dunkerley D.L. y Brown K.J. (1999). Banded vegetation near Broken Hill, Australia: significance of surface roughness and soil physical properties. Catena, 37, 75-88.

Galle S., Ehrmann M. y Peugeot C. (1999). Water balance in a banded vegetation pattern: A case study of tiger bush in western Niger. Catena, 37, 197-216.

Ludwig J., Tongway D., Freudenberger D., Noble J. y Hodgkinson K. (eds.) (1997). Landscape ecology, function and management: principles from Australia's rangelands. CSIRO Publishing, Melbourne, Australia.

Ludwig J.A., Tongway D.J. y Marsden S.G. (1999). Stripes, strands or stipples: modelling the influence of three landscape banding patterns on resource capture and productivity in semi-arid woodlands, Australia. Catena, 37, 257-273.

Ludwig J.A., Wilcox B.P., Breshears D.D., Tongway D.J. y Imeson A.C. (2005). Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. Ecology, 86, 288-297.

McIvor J.G., Williams J. y Gardener C.J. (1995). Pasture management influences runoff and soil movement in the semiarid tropics. Australian Journal of Experimental Agriculture, 35, 55-65.

Moreno-de las Heras M. (2009a). Development of soil physical structure and biological functionality in mining spoils affected by soil erosion in a Mediterranean-Continental environment. Geoderma, 149, 249-256.

Moreno-de las Heras M. (2009b). Efectos ecológicos de la erosión en laderas derivadas de la restauración de la minería a cielo abierto. En: Departamento de Ecología. Universidad de Alcalá Alcalá de Henares.

Moreno-de las Heras M., Merino-Martín L. y Nicolau J.M. (2009). Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. Catena, 77, 39-47.

Moreno-de las Heras M., Nicolau J.M. y Espigares T. (2008). Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment. Ecological Engineering, 34, 168-178.

Moreno-de las Heras M., Nicolau J.M., Merino-Martin L. y Wilcox B.P. (2010). Plot-scale effects on runoff and erosion along a slope degradation gradient. Water Resources Research, 46, 12.

Nicolau J.M. (2002). Runoff generation and routing on artificial slopes in a Mediterranean-continental environment: the Teruel coalfield, Spain. Hydrological Processes, 16, 631-647.

Puigdefábregas J. (2005). The role of vegetation patterns in structuring runoff and sediment fluxes in drylands. Earth Surface Processes and Landforms, 30, 133-147.

Puigdefábregas J., Sole A., Gutierrez L., del Barrio G. y Boer M. (1999). Scales and processes of water and sediment redistribution in drylands: results from the Rambla Honda field site in Southeast Spain. Earth-Science Reviews, 48, 39-70.

Scanlan J., Pressland A. y Myles D. (1996). Run-Off and Soil Movement on Mid-Slopes in North-East Queensland [Australia] Grazed Woodlands. The Rangeland Journal, 18, 33-46.

Seghieri J. y Galle S. (1999). Run-on contribution to a Sahelian two-phase mosaic system: Soil water regime and vegetation life cycles. Acta Oecologica-International Journal of Ecology, 20, 209-217.

Valentin C., d'Herbes J.M. y Poesen J. (1999). Soil and water components of banded vegetation patterns. Catena, 37, 1-24.

Wainwright J., Parsons A.J., Schlesinger W.H. y Abrahams A.D. (2002). Hydrology-vegetation interactions in areas of discontinuous flow on a semi-arid bajada, Southern New Mexico. Journal of Arid Environments, 51, 319-338.

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

Capítulo 8

A Juan, Manuel, Diego Martín y Joan Manuel

y bailan y bailan
todo el día bailan
y nunca paran
nunca descansan
luego se acarician
llenos de caricias
y nunca paran
nunca descansan
y se sienten tiernos
todo el día tiernos
y nunca paran
nunca descansan

Pedro Guerra

Capítulo 8

Conclusiones generales

A escala regional

1. La composición específica de las comunidades vegetales a escala de mancha se corresponde directamente con tres tipos fundamentales de comportamiento ecohidrológico: fuentes, sumideros y transición. De esta forma, la interacción escorrentía-vegetación configura a escala de mancha "unidades ecohidrológicas".

2. Las interrelaciones ecohidrológicas se relacionan con el grado de perturbación de una ladera, existiendo dos tendencias dirigidas hacia una menor intensidad de las interacciones. Una tendencia caracterizada por el establecimiento de la vegetación y como consecuencia, disminución del ratio fuente/sumidero y otra caracterizada por procesos de formación de regueros, en los que la redistribución del agua entre fuentes y sumideros es sorteada, y éstos dirigen los procesos ecohidrológicos realizando una evacuación eficiente de la escorrentía.

A escala de ladera

3. La circulación y generación de escorrentía en laderas restauradas sometidas a los primeros estadios de sucesión y con un régimen hidrológico de dominio de la escorrentía laminar y/o reguerización discontinua mostraron un patrón en fuentes y sumideros de escorrentía.

4. Dos clases de sumideros de escorrentía fueron identificados: "sumideros profundos" (relacionados con la aparición de *Genista scorpius*) y "sumideros en derramaderos de escorrentía" (relacionados con la aparición de comunidades de herbáceas caracterizadas por *Lolium perenne* o por *Brachypodium retusum*). Ambos se relacionan con los dos posibles mecanismos de generación de mosaicos en manchas de laderas semiáridas: patrones dirigidos por la geomorfología (laderas 1 y 2, donde se desarrollan los sumideros en derramaderos de escorrentía) y patrones dirigidos por la acción de la vegetación (ladera 3, con sumideros profundos). En este caso, los patrones de generación de mosaicos están asociados a un umbral de volumen de escorrentía circulante, de manera que volúmenes altos generan patrones dirigidos o condicionados por la geomorfología, mientras que volúmenes bajos de escorrentía propician la aparición de patrones dirigidos por la vegetación.

5. El volumen de escorrentía que discurre por la ladera controla la "diversidad hidrológica". Cuando el volumen de escorrentía aumenta, la diversidad hidrológica disminuye, desarrollando comportamientos hidrológicos muy contrastados (fuentes extremas y sumideros). Cuando se presentan bajos volúmenes de escorrentía circulante, la diversidad hidrológica aumenta, los comportamientos hidrológicos se atemperan y se observa un cambio cualitativo, la aparición de un "sumidero profundo" (*Genista scorpius*).

6. Se ha demostrado que el marco conceptual TTRP (Trigger-Transfer-Reserve-Pulse) desarrollado para laderas semiáridas naturales es aplicable a laderas restauradas de ambientes mediterráneos en los primeros estadios de sucesión. Se ha comprobado que los sumideros realizan una obstrucción del flujo de escorrentía, lo que proporciona un mayor contenido de humedad, que a su vez favorece el rendimiento de las comunidades vegetales (riqueza florística y del banco de semillas, estrés hídrico, producción de biomasa), y que en algunos casos (ladera 3) propicia la aparición de fenómenos de retroalimentación entre la vegetación y la hidrología favorecedores del desarrollo vegetal.

7. En laderas restauradas de ambientes semiáridos, donde el agua es uno de los factores limitantes, la escorrentía es un factor director de la sucesión ecológica. Este factor dirige la colonización y el establecimiento de la vegetación mediante el incremento del déficit hídrico.

8. Se identificó un umbral de volumen de escorrentía circulante (entre las laderas 2 y 3) que determina el predominio del control abiótico de las interrelaciones ecohidrológicas (por parte de la hidrología) o del control biótico del agua (por parte de la vegetación). El control biótico del agua aumenta según disminuye la escorrentía por el efecto de las manchas de matorral de *Genista scorpius*, que crea islas de intensa actividad hidrológica.

Recomendaciones para la restauración de laderas en ambientes semiáridos

9. El diseño topográfico de las laderas debe evitar la existencia de áreas-fuente de escorrentía en las cabeceras (cejas, convexidades, conexiones con pistas superiores) que incrementan el volumen de escorrentía circulante, reduciendo las probabilidades del control biológico del ciclo del agua y, por tanto, del desarrollo de la vegetación. Se recomiendan los perfiles de ladera cóncavos frente a los convexo-rectilíneo-cóncavos.

10. La introducción de especies capaces de iniciar procesos de "nucleación" y de dirigir el sistema restaurado hacia un control biótico del agua puede ser fundamental en estos ambientes (en nuestro caso *Genista scorpius*). Asimismo, puede resultar también una buena aplicación práctica el redirigir la escorrentía hacia los núcleos en los que se introduzcan especies clave que se puedan beneficiar de esta fuente de agua adicional y dirigir el proceso sucesional hacia la "agradación".

11. El hecho de que las laderas restauradas se caractericen por altos volúmenes de escorrentía y que ésta haya sido identificada como un factor director de la sucesión de estos ambientes, sugiere el denominado "manejo experto de la escorrentía" como un elemento fundamental para el logro de los objetivos de las restauraciones.

Anexo fotográfico

Por tierras de España

El hombre de estos campos que incendia los pinares
y su despojo aguarda como botín de guerra,
antaño hubo raído los negros encinares,
talado los robustos robledos de la sierra.

Hoy ve sus pobres hijos huyendo de sus lares;
la tempestad llevarse los limos de la tierra
por los sagrados ríos hacia los anchos mares;
y en páramos malditos trabaja, sufre y yerra,
…
Veréis llanuras bélicas y páramos de asceta
no fue por estos campos el bíblico jardín ;
son tierras para el águila, un trozo de planeta
por donde cruza errante la sombra de Caín.

Antonio Machado

Fig. 1. Acumulación de escorrentía procedente de laderas restauradas y cunetas en las plataformas del Cerro del Moral.

Fig. 2. Patrón de distribución de la vegetación de la ladera 1. Manchas *Lolium* en conos de deyección de regueros y *Santolina* en interregueros.

Estación experimental de El Moral (capítulos 4, 5 y 6):

Escala de ladera

Figura 3. Parcela de escorrentía instalada en la ladera 2, con manchas *Brachypodium* en zonas de derramaderos de escorrentía y detalle de colector a pie de ladera.

Estación experimental de El Moral (capítulos 4, 5 y 6):

Escala de mancha

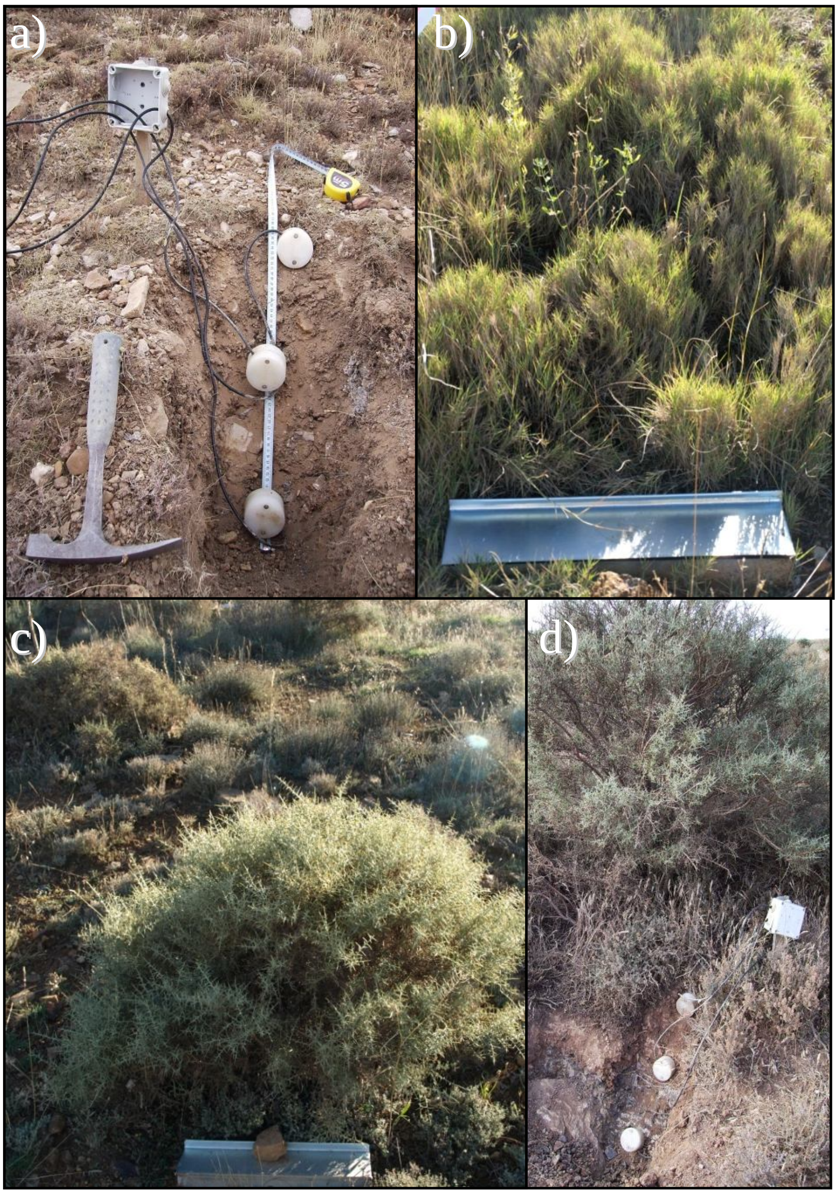


Figura 4. Detalle de sondas TDR (Time Domain Reflectometry) instaladas a tres profundidades (5, 25 y 50 cm) en manchas *Thymus* (a) y *Genista* (d) y cajas gerlachs en manchas *Brachypodium* (b) y *Genista* (c).

Experimento de exclusión de escorrentía (capítulo 6):

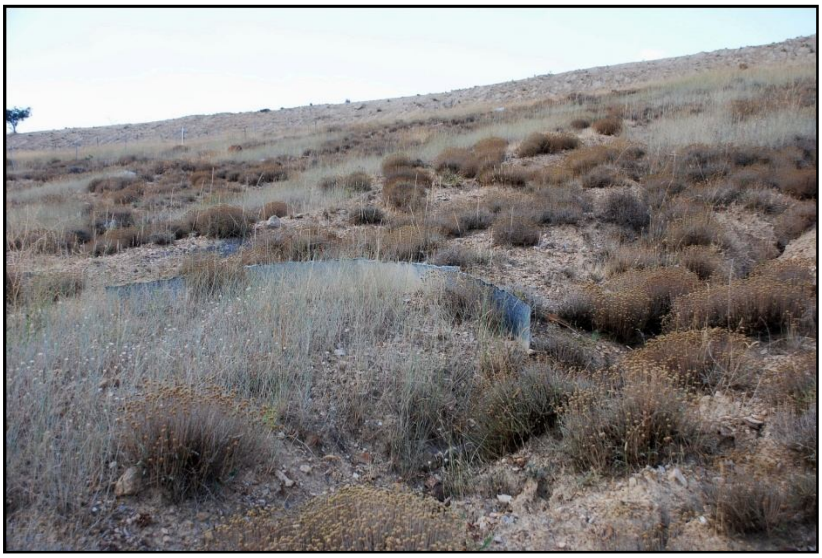


Figura 5. Experimento de exclusión de escorrentía en mancha *Lolium* (ladera 1) .

Figura 6. Experimento de exclusión de escorrentía en mancha *Brachypodium* (ladera 2).

Experimento de exclusión de escorrentía (continuación):

Figura 7. Experimento de exclusión de escorrentía en mancha *Genista* (ladera 3) .

Le he preguntado a mi sombra
a ver como ando para reírme,
mientras el llanto, con voz de templo,
rompe en la sala
regando el tiempo.

Mi sombra dice que reírse
es ver los llantos como mi llanto,
y me he callado, desesperado
y escucho entonces:
la tierra llorar.

La era está pariendo un corazón,
no puede más, se muere de dolor
y hay que acudir corriendo
pues se cae el porvenir
en cualquier selva del mundo,
en cualquier calle.

Debo dejar la casa y el sillón,
y hay que quemar el cielo si es preciso
por vivir,
por cualquier hombre del mundo,
por cualquier casa,
por cualquier casa.

La era está pariendo un corazón
Silvio Rodríguez (Cuba) - 1968