Tesis Doctoral

Efectos ecológicos de la erosión en laderas derivadas de la minería del carbón a cielo abierto

Mariano Moreno de las Heras 2009

Universidad de Alcalá

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Alcalá de Henares, a 19 de enero de dos mil nueve.

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Alcalá de Henares, a 19 de enero de dos mil nueve.

Dr. Miguel Ángel Rodríguez Fernández
DIRECTOR DEL DEPARTAMENTO

A mis padres,
Mariano y Antonia

The threat of nuclear weapons and man's ability to destroy the environment are really alarming. And yet there are other almost imperceptible changes - I am thinking of the exhaustion of our natural resources, and especially of soil erosion - and these are perhaps more dangerous still, because once we begin to feel their repercussions it will be too late.

Tenzin Gyatso
XIV Dalai Lama

Agradecimientos

Largo ha sido el camino que finalmente me ha llevado hasta aquí, hasta estas líneas. Si bien, la firma de esta tesis doctoral refleja en apariencia una labor autónoma e individual, no es menos cierto que ésta no hubiera sido posible sin la colaboración activa y el apoyo de un importante número de personas. Quisiera expresar en estas líneas mi profundo agradecimiento a todas las personas que de un modo u otro me han acompañado a lo largo esta aventura.

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M.

Resumen

La minería a cielo abierto del carbón representa una actividad con una elevada capacidad de transformación del territorio. La restauración de las áreas afectadas por esta actividad es una tarea compleja, dado el nivel de alteración de los ecosistemas afectados y los múltiples factores involucrados en el desarrollo de los nuevos ecosistemas restaurados. Un fenómeno característico que puede limitar el desarrollo de estos ecosistemas artificiales es el desarrollo temprano de procesos intensos de erosión hídrica superficial con formación de redes de regueros. Al mismo tiempo, en ambientes con restricciones hídricas uno de los principales mecanismos por los que la erosión puede interferir en el desarrollo de la vegetación es la reducción de la disponibilidad de agua para las plantas. El objetivo de esta tesis doctoral es determinar los efectos de la erosión sobre la disponibilidad y distribución del agua en el suelo y sus consecuencias sobre la dinámica de las comunidades vegetales y el desarrollo de los procesos biológicos del suelo en ecosistemas restaurados de ladera derivados de la minería a cielo abierto del carbón de ambiente mediterráneo-continental.

Para abordar este objetivo se ha llevado a cabo un primer trabajo a escala regional ("la cuenca lignitífera de Teruel") en el que se han identificado los grandes patrones de sucesión ecológica en laderas restauradas del territorio. Éste ha puesto de manifiesto un panorama complejo de evolución temporal, caracterizado por el desarrollo de múltiples trayectorias sucesionales. Los procesos de erosión en regueros (desencadenados en general por aportes de escorrentía superficial desde la cabecera de las laderas restauradas y estructuras mineras superiores a éstas) constituyen a escala regional una fuerza directriz de las trayectorias de degradación en las comunidades vegetales restauradas, conduciendo a la formación de comunidades muy ralas y poco diversas básicamente constituidas por unos pocos individuos de una especie perenne introducida mediante siembra: *Medicago sativa*.

La identificación de este problema dio lugar a su estudio detallado a partir de un conjunto de trabajos llevados a cabo en cinco laderas restauradas del "Área experimental de Utrillas". Estas laderas fueron construidas con características topográficas, edáficas y de revegetación muy similares, sin embargo han desarrollado procesos de erosión en regueros de diferente intensidad (de 0 a 70 t ha^{-1} año^{-1}) a causa de la presencia de áreas de contribución de escorrentía de diferente tamaño en sus cabeceras. Estos trabajos centran el análisis sobre: (a) la dinámica estacional de generación y (b) la dinámica espacial de circulación de los flujos de escorrentía y sedimentos, (c) la disponibilidad y distribución espacial de la humedad en el suelo, (d) la estructura y dinámica de la vegetación, y (e) el desarrollo de las condiciones físicas y funcionalidad biológica del suelo.

Los resultados obtenidos reflejan la importancia clave que los procesos intensos de erosión con formación de redes de regueros tienen sobre la organización de los ecosistemas restaurados a través de la reducción de los recursos hídricos disponibles para el desarrollo de la vegetación. Así, el desarrollo de las redes de regueros condiciona el patrón de circulación de los flujos superficiales, maximizando las pérdidas de agua de las laderas en forma de escorrentía superficial. Simultáneamente, estas redes condicionan la distribución espacial de la humedad en el suelo, concentrando los recursos hídricos en los regueros, áreas donde el desarrollo de la vegetación se ve imposibilitado por el impacto mecánico de los flujos concentrados que circulan por éstos. Como resultado directo se produce una drástica reducción de carácter no lineal en la cantidad de agua disponible para las plantas asociada a las tasas de erosión y, en consecuencia, un aumento significativo de los niveles de estrés hídrico soportados por la vegetación y el bloqueo de los procesos de colonización vegetal. La suma de estos efectos produce a escala de ladera reducciones paralelas de carácter exponencial en la diversidad y biomasa de la vegetación, produciendo una simplificación drástica de las comunidades restauradas. El extremo de mayor simplificación está constituido por las comunidades vegetales ralas dominadas por individuos de *M. sativa*, cuya organización espacial se encuentra condicionada por los patrones de distribución de la humedad del suelo y perturbación mecánica que generan las redes de regueros. Las caídas de carácter exponencial observadas en los atributos estructurales de la vegetación (biomasa y riqueza) son mantenidas en los niveles asociados de desarrollo de la estructura física (estabilidad de los agregados del suelo) y la funcionalidad biológica del suelo (actividad microbiana y actividad de diferentes hidrolasas que regulan el reciclado de los elementos del suelo).

La naturaleza no lineal de las relaciones determinadas entre los procesos de erosión y la vegetación ha permitido identificar unos umbrales críticos precisos que determinan trayectorias de evolución distintas en los ecosistemas artificiales de ladera estudiados: (a) con niveles de desarrollo de la cubierta vegetal inferiores al 30% y tasas de erosión en regueros superiores a 20 t ha^{-1} $año^{-1}$ se produce la transición hacia la formación de las comunidades vegetales simples, ralas y poco productivas dominadas por *M. sativa*; (b) con niveles de desarrollo de la cubierta superiores al 50% y tasas de erosión en regueros inferiores a 5 t ha^{-1} $año^{-1}$ la dinámica de la vegetación no se encuentra condicionada por los procesos de erosión y su evolución responde a otros factores y procesos, fundamentalmente de tipo biótico.

Palabras clave: disponibilidad de agua, erosión en regueros, escorrentía, clima mediterráneo-continental, restauración minera, Teruel, umbral, dinámica de la vegetación.

Abstract

Opencast coal-mining is considered one of the most dramatic human disturbances in terrestrial ecosystems. Mining reclamation represents a complex task on account of the drastic alterations caused by this activity and of the multiple factors involved in the recovery of the disturbed ecosystems. Reclaimed lands from opencast mining are particularly vulnerable to the effects of accelerated soil erosion processes, especially when these processes lead to the formation of rill networks. One of the effects caused by soil erosion is the reduction of water availability for plants, which can noticeably constrain the development of vegetation and hence reclamation success in water-limited environments. The objective of this dissertation is to analyse the effects of soil erosion in the availability and distribution of soil moisture, as well as the consequences of these effects in vegetation dynamics, and the development of soil biological processes in reclaimed slopes derived from opencast coal-mining in a Mediterranean-Continental environment.

Firstly, a regional scale study was carried out to identify the pattern and factors controlling vegetation succession in reclaimed slopes of "the Teruel coalfield" (central-eastern Spain). This work revealed a complex pattern characterised by the development of multiple successional trajectories. Rill erosion processes (triggered in general by run-on fluxes coming from the top of reclaimed slopes and up-slope mining structures) represent at this regional scale a significant driving force for restored vegetation dynamics, leading to a very sparse and simple community essentially constituted by a few individuals of *Medicago sativa* (a sown perennial legume).

This erosion-related trend was analysed deeply in a series of specific studies carried out in five reclaimed slopes located in the "Utrillas field site". In spite of having very similar reclamation treatments and initial features, these slopes have developed rill erosion processes of different intensity (from 0 to 70 t ha^{-1} $year^{-1}$) owing to the presence of overland flow contributing areas of different size up-slope. The objectives of these works are the analysis of: (a) the temporal dynamics of generation and (b) the spatial dynamics of circulation of runoff and sediment fluxes, (c) the availability and spatial distribution of soil moisture, (d) the dynamics and structure of vegetation, and (e) the development of the physical structure and biological functionality of the soil.

The results obtained highlight the role of rill erosion processes in the development of reclaimed ecosystems by diminishing the availability of water for vegetation. In this way, the circulation of surface flows is conditioned by the development of rill networks, thus maximizing the loss of water resources at the slope scale by surface runoff. Additionally, these channelling networks organise the spatial distribution of soil moisture, since they concentrate water resources around rill beds (areas where vegetation development is prevented by the mechanical disturbance associated to the concentrated runoff flows). As a result, the availability of water resources for plants is drastically reduced through a non-linear erosion-related function. Consequently, vegetation performance and dynamics are affected by the increase of drought stress and the inhibition of natural plant colonisation. All these effects together are reflected in ecosystem structure at the slope scale by an exponential decline of species richness and vegetation biomass, showing a drastic simplification of plant communities related to soil erosion rates. The most simplified state is represented by the sparse and poor communities dominated by *M. sativa* plants, in which case their spatial organisation is ruled by the distribution of soil moisture and mechanical disturbance derived from rill networks. In addition to the identified effects on vegetation, other variables concerning the development of both soil physical structure (soil aggregates stability) and soil biological functionality (the activity of soil microorganisms and a set of soil hydrolases basic for the nutrient cycling) also showed exponential decreases associated to soil erosion rates.

A critical threshold operating in the studied reclaimed ecosystems has been determined on the basis of the non-linear nature of the erosion-vegetation relationships identified: (a) vegetation in reclaimed slopes with under 30% vegetation cover and rill erosion rates over 20 t ha^{-1} $year^{-1}$ regress to the low diversity and sparse plant communities dominated by *M. sativa*; (b) when vegetation cover is over 50% and rill erosion rates remain below 5 t ha^{-1} $year^{-1}$, vegetation dynamics of these constructed slopes is not conditioned by soil erosion processes and ecosystem dynamics rely on other factors and processes.

Key words: Mediterranean-Continental climate, mining reclamation, overland flow, rill erosion, Teruel, threshold, vegetation dynamics, water availability.

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Capítulo 1

Capítulo 1

Introducción general

Antecedentes y estado actual del tema

La minería del carbón en el contexto del Cambio Global de la Biosfera

A escala global, Hooke (2000) ha estimado que el movimiento de tierras generado por la construcción de viviendas, carreteras y la explotación de recursos minerales (Imagen 1.1) se sitúa entre 30 y 45 Gt al año. Estos valores son comparables o incluso superiores a los movimientos causados por los agentes naturales, como el transporte a larga distancia de sedimentos por los ríos (24 Gt $año^{-1}$), la acción de los glaciares (4,3 Gt $año^{-1}$), los procesos de ladera (0,6 Gt $año^{-1}$) o la formación de montañas y volcanes por la tectónica en los continentes y en los océanos (14 y 30 Gt $año^{-1}$). Por tanto, se puede afirmar que la especie humana representa en la actualidad un agente geomorfológico de primer orden, formando parte de los síndromes característicos del Cambio Global de la Biosfera.

Las actividaes mineras (Imagen 1.1) representan en conjunto la mitad de los movimientos de tierras generados por el ser humano (Hooke, 1994); en superficie estas actividades afectan de forma directa a cerca del 1% de la superficie emergida de la Tierra (Walker y del Moral, 2003). Cabe destacar entre los diferentes tipos de actividades extractivas la minería del carbón, recurso mineral básico tanto en la estructura de la demanda energética global como de la española, cuya explotación ha experimentado en los dos siglos pasados un espectacular crecimiento que en la actualidad se mantiene a ritmo acelerado (Cuadro 1.1).

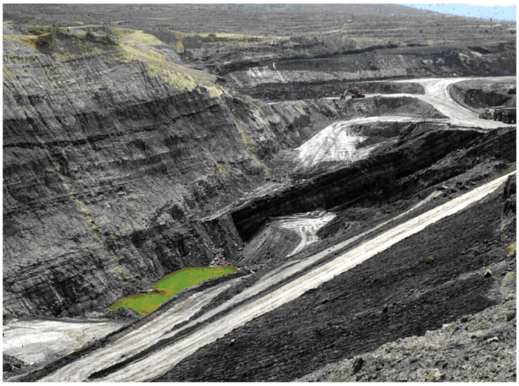

Imagen 1.1. Explotación de recursos minerales (carbón) a cielo abierto. El desarrollo acelerado de estas actividades reflejan la emergencia, dentro de los síndromes característicos del Cambio Global de la Biosfera, del ser humano como agente geológico de primer orden.

Cuadro 1.1. Explotación del carbón como recurso energético clave

El incremento de la población humana (estimada para septiembre de 2008 en 6700 millones de personas; WPC) y los avances tecnológicos han tenido como consecuencia un continuo aumento de las tasas de extracción de recursos minerales. Este efecto es especialmente visible en el caso de los recursos energéticos como el carbón (Figura 1.1), que alimenta el 25% de la producción de energía a nivel mundial (IEA, 2006). Dos hechos de considerable importancia produjeron cambios notables en las tasas históricas de extracción del carbón: la industrialización, a partir de finales del siglo XIX, y el comienzo de la minería a cielo abierto gracias al desarrollo de maquinaria con elevada capacidad de carga y movilización de tierra, tras la Segunda Guerra Mundial. En la actualidad las tasas de extracción de carbón están experimentando una nueva aceleración, en este caso impulsada por el desarrollo de las economías asiáticas (en especial China e India), que ya acaparan el 45% de la demanda global (IEA, 2006). Las previsiones futuras de consumo mundial estiman un incremento del 73% de la demanda de carbón entre los años 2005 y 2030 impulsado por el desarrollo de las economías emergentes asiáticas y el panorama alcista del precio del petróleo (IEA, 2007).

El carbón es una fuente energética fundamental en la estructura del sistema eléctrico español. Así, la combustión de este recurso en centrales térmicas ha sostenido a lo largo de las últimas dos décadas cuotas que representan entre el 25 y el 35% del total de la producción eléctrica nacional (REE, 2006; 2008). No obstante, la extracción de carbón en España se encuentra en franca decadencia, registrando una disminución del 30% a lo largo de la última década, para alcanzar en el año 2006 tasas de extracción en torno a 18 Mt (Figura 1.2). Efectivamente, aunque la cantidad de reservas de carbones nacionales es muy importante (4875 Mt; IGME, 1985), la dificultad que presenta su extracción a precios competitivos en unos casos, y las limitaciones a las emisiones contaminantes de azufre que impone la normativa ambiental europea en otros, hacen que su explotación se encuentre en fase de regresión (Bustillo y López, 1996). El nuevo plan nacional del carbón 2006-2012 prevé una mayor contracción de la producción de carbones nacionales, alcanzando para el horizonte 2012 un objetivo de extracción de 9 Mt. La explotación nacional se mantendría así únicamente en las minas a cielo abierto y subterráneas más rentables, consideradas de carácter estratégico. El escenario futuro se completa con la tendencia creciente a la importación de carbones de alta calidad (alto contenido energético y bajos contenidos de azufre), que ya en el año 2006 representaron el 62% de los 38 Mt de carbón consumidos por el sector eléctrico español (IGME, 2007).

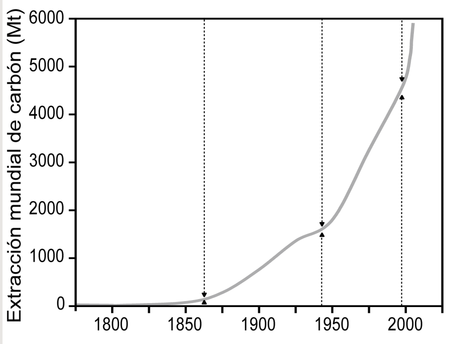

Figura 1.1. Evolución histórica mundial de la extracción de carbón. Las líneas discontinuas verticales representadas marcan los diferentes momentos históricos en los que se ha producido una aceleración en las tasas de extracción del carbón. Datos de tomados de McNeil (2000) y Hetherington et al. (2007).

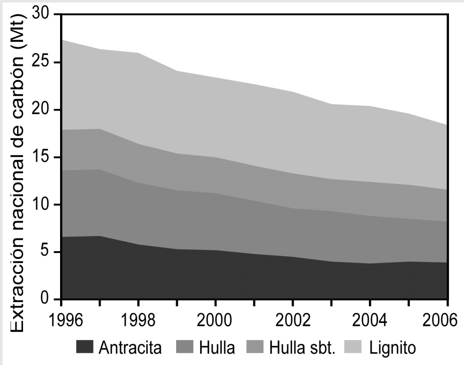

Figura 1.2. Evolución de la extracción de carbón en España. Datos tomados de IGME (2002; 2007).

Dinámica de los ambientes restaurados mineros: determinismo y contingencia

Desde las ciencias geomorfológicas se ha elaborado una particular forma de clasificar las áreas degradadas terrestres: en función de la "profundidad de alteración" en la columna formada por el sistema vegetación-suelo-subsuelo (Toy y Hadley, 1987). En el caso de la minería del carbón a cielo abierto las alteraciones producidas inciden sobre todos los compartimentos del sistema (Cuadro 1.2), debiendo acometer actuaciones de restitución en el relieve, suelo y vegetación para la reconstrucción de un ecosistema funcional (Gómez-Sal y Nicolau, 1999). Los profundos cambios producidos hacen de la restauración de estas áreas alteradas una tarea muy compleja (Whisenant, 2002).

Aunque los métodos y técnicas de restauración minera han experimentado un amplio desarrollo durante las últimas tres décadas, los fallos en las restauraciones realizadas han sido frecuentes, la calidad de los terrenos restaurados en numerosos casos no es aceptable y su degradación progresiva representa un problema ambiental de consideración (Haigh, 2000). En definitiva, existen aspectos pobremente comprendidos que demandan el desarrollo de una base científica sólida que sirva de fundamento para la práctica de las restauraciones (Plass, 2000; Halle y Fattorini, 2004; Méndez *et al*., 2008).

De la mano del estudio de los procesos de colonización y sucesión ecológica, se están realizando numerosos esfuerzos para identificar y comprender los factores limitantes y las fuerzas directoras que pueden condicionar el resultado final de las restauraciones (Young *et al*., 2005; Walker y del Moral, 2009). Dos grandes grupos de factores modulan los procesos de desarrollo de estos ambientes restaurados: por un lado factores deterministas y por otro factores contingentes (de carácter circunstancial), que pueden condicionar de forma impredecible los resultados de las restauraciones (Pickett *et al*., 2001; Nicolau y Moreno-de las Heras, 2005). El clima y las condiciones de partida de los sistemas restaurados juegan un papel clave dentro del marco determinista (Davy, 2002). Así, el clima constituye un primer factor de control, condicionando tanto la disponibilidad de recursos hídricos como el régimen de temperaturas para la emergencia y desarrollo de las plantas (Greller, 1974; Otto *et al*., 2006). Otros factores locales relacionados con el diseño geomorfológico, las características físico-químicas del suelo (textura, compactación, disponibilidad de nutrientes, pH, toxicidad), así como la disponibilidad de semillas y especies vegetales empleadas, tienen un papel clave para el éxito de las restauraciones (Bradshaw, 1997; Wali, 1999; Hancock *et al*., 2003; Prach *et al*., 2007; Tormo *et al*., 2006).

Los ecosistemas restaurados son sistemas abiertos, por lo que los ecosistemas del entorno influyen notablemente en su evolución (Zamora, 2002), principalmente a través de los flujos de propágulos y de escorrentía desde áreas externas, muy condicionados por la distancia y el estado de conservación de las áreas circundantes. Otro tipo de influencias están asociadas al régimen de perturbaciones naturales (plagas, incendios, precipitaciones de alta intensidad, ciclos de sequía, etc.). Finalmente, hay que reseñar la importancia de la influencia antrópica

Cuadro 1.2. Minería a cielo abiero del carbón: explotación, impactos generados y regulación

La minería a cielo abierto del carbón generalmente se desarrolla por el método de transferencia entre paneles (ITGME, 1996; Abril y Molina, 1997). Este sistema se basa en la apertura de un hueco que progresa a lo largo del área donde se extiende el yacimiento, rellenándose conforme el material es extraído. Consta de las siguientes fases:

a) Retirada de la vegetación y acopio de los suelos ("tierra vegetal") de la superficie de los terrenos que van a ser explotados.

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

c) Vertido de los materiales estériles extraídos en una zona externa, conformando una escombrera exterior.

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

e) Estos pasos se repiten con la apertura de un nuevo panel a continuación del anterior, siguiendo la dirección de las capas del yacimiento. Los nuevos materiales estériles extraídos se depositan en el interior del hueco dejado por el anterior panel, conformando una escombrera interior.

En un esquema idealizado (Figura 1.3a), la organización final quedaría constituida por una escombrera exterior de grandes dimensiones y una serie de escombreras interiores de menor tamaño, seguidas de un hueco final.

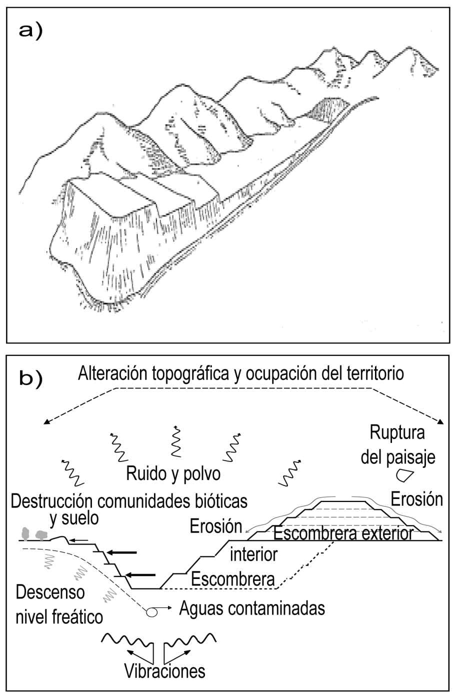

Figura 1.3. (a) Configuración final típica de una explotación de carbón a cielo abierto mediante el método de cortas con transferencia de paneles; tomado de Nicolau y Ruiz (1986). (b) Impactos asociados a la explotación a cielo abierto del carbón; tomado con modificaciones de Artieda y Agudo (1984).

La restauración se acomete de forma simultánea a la explotación y suele constar de las siguientes actuaciones (Nicolau y Moreno-de las Heras, 2006; Imagen 1.2):

a) 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.

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

d) Revegetación. Las operaciones más simples están constituidas por la siembra (por métodos manuales o hidrosiembra) de mezclas de semillas de especies herbáceas.

La minería a cielo abierto del carbón constituye una actividad que impacta con gran intensidad sobre una amplia gama de variables ambientales (Figura 1.3b). Estos impactos se pueden sintetizar 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, voladuras, 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: La ocupación directa de las extensiones que son explotadas 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.

El desarrollo de la minería a cielo abierto provocó en España una fuerte preocupación social en los años 80, que desembocó en la elaboración de un marco legislativo específico. A nivel estatal se dictó en 1982 una norma genérica (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 entrada 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 la Ley 6/2001 de 8 de mayo). 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.

Imagen 1.2. Vista de las diferentes fases comprometidas en la restauración de las áreas alteradas por la minería a cielo abierto del carbón: conformación de la topografía, extensión de tierra vegetal y revegetación.

sobre la evolución de los ecosistemas restaurados, especialmente sobre aquellos artificialmente construidos, como es el caso de los ambientes mineros. En este aspecto hay que destacar fenómenos de carácter contingente como los errores de ejecución en las operaciones de restauración, la introducción no prevista de actividades y usos del territorio, así como cambios en la legislación aplicable (Wiegleb y Felinks, 2001; Simpson, 2002; Nicolau y Moreno-de las Heras, 2005).

La interacción de todos estos factores hace posible la presencia de trayectorias alternativas de evolución en los ecosistemas restaurados (Lockwood y Samuels, 2004). Así, el abanico posible de resultados puede ser muy amplio, incluyendo desde diferentes trayectorias de mejora de las condiciones y desarrollo del ecosistema restaurado hasta trayectorias de degradación continua (Figura 1.4).

Impacto de la erosión sobre la evolución de las áreas restauradas mineras

Uno de los principales fenómenos ligados al desarrollo de trayectorias de degradación en ambientes restaurados es el desencadenamiento de procesos acelerados de erosión, a los que son especialmente sensibles las laderas mineras (Rubio-Montoya y Brown, 1984; Haigh, 2000; Nicolau y Asensio, 2000).

Efectivamente, los substratos empleados para la restauración de estas superficies (frecuentemente asociados con bajos niveles de desarrollo edáfico) suelen estar caracterizados por una baja capacidad de infiltración y una alta erodibilidad (Imagen 1.3), muy especialmente cuando no se ha realizado extensión de tierra vegetal (Ward *et al.*, 1983; Ritter, 1992; Guebert y Gardner, 2001). Sumado a este hecho, pueden concurrir otros problemas derivados de errores en el diseño geomorfológico que favorezcan la acumulación de escorrentía (la presencia de áreas de contribución

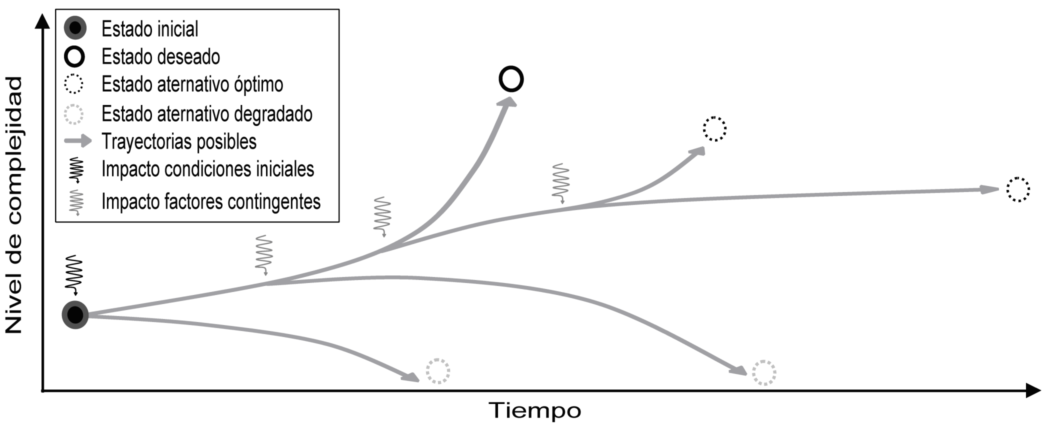

Figura 1.4. Diagrama conceptual que representa las múltiples trayectorias de evolución que puede seguir un ecosistema restaurado como consecuencia del impacto de los múltiples factores (deterministas y contingentes) involucrados. Tomado con modificaciones de Lockwood y Samuels (2004).

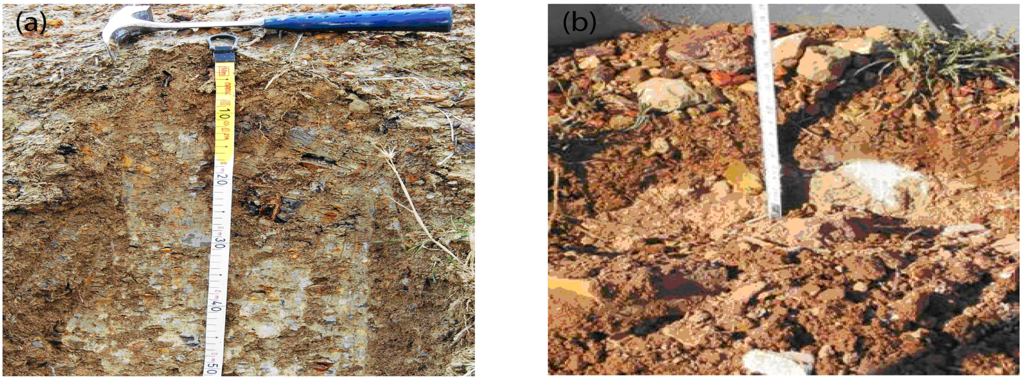

Imagen 1.3. Los suelos mineros jóvenes están en general caracterizados por una falta muy considerable de desarrollo edáfico. En la figura (a) se puede observar el perfil de un suelo minero joven, sin diferenciación edáfica y con estructura masiva. La figura (b) representa la baja capacidad de infiltración (2-3 cm de profundidad del frente de humectación tras una lluvia simulada de 40 minutos a 63 mm h^{-1}) que caracteriza a estos suelos masivos.

de escorrentía en la cabecera de las laderas restauradas o la conexión directa de estas partes con flujos de escorrentía procedentes de bermas, canales o plataformas superiores; Imagen 1.4) o que favorezcan la aceleración de los flujos superficiales (excesiva pendiente de la ladera), pudiendo provocar el desencadenamiento temprano de procesos de erosión intensos (Evans, 1997; Nicolau, 2003; Hancock y Willgoose, 2004). En sus versiones más severas, cuando estos procesos dan lugar a la formación de redes integradas de regueros, la magnitud de la erosión alcanzada en los terrenos restaurados mineros puede llegar a ser muy alta (Tabla 1.1), comparable a las registradas en taludes de carretera y campos de cárcavas o baldíos (Estalrich *et al*., 1997; Andrés y Jorba, 2000; Desir y Marín, 2007; Nadal-Romero *et al*., 2008).

El desencadenamiento de procesos acelerados de erosión hídrica superficial en los ambientes restaurados mineros tiene consecuencias dramáticas tanto para los sistemas del entorno (asociadas a la emisión de flujos de escorrentía y sedimentos), como para el propio sistema erosionado (Bender *et al*., 2000; Sawatski *et al*., 2000; Nicolau, 2003). A este último nivel, los efectos de los procesos de erosión hídrica superficial se manifiestan mediante la degradación de la vegetación restaurada y el bloqueo de los procesos de colonización (Haigh, 1992; Nicolau y Asensio, 2000; Kapolka y Dollhopf, 2001; Wang *et al*., 2007).

Relaciones entre la vegetación y los procesos de erosión hídrica superficial

Tradicionalmente, el análisis de los impactos causados por la erosión hídrica superficial sobre la vegetación ha interpretado este proceso como un agente de explotación abiótico. De este modo, la erosión es identificada como agente responsable del desarraigo de las plantas y la pérdida de semillas (Chambers y McMahon, 1994; Cerdà y García-Fayos, 1997; Chambers, 2000), así como de la reducción de la disponibilidad de agua y nutrientes en el suelo (Pimentel *et al*., 1995; Zheng, 2005).

Por otro lado, es ampliamente reconocido el papel de control que la vegetación ejerce sobre

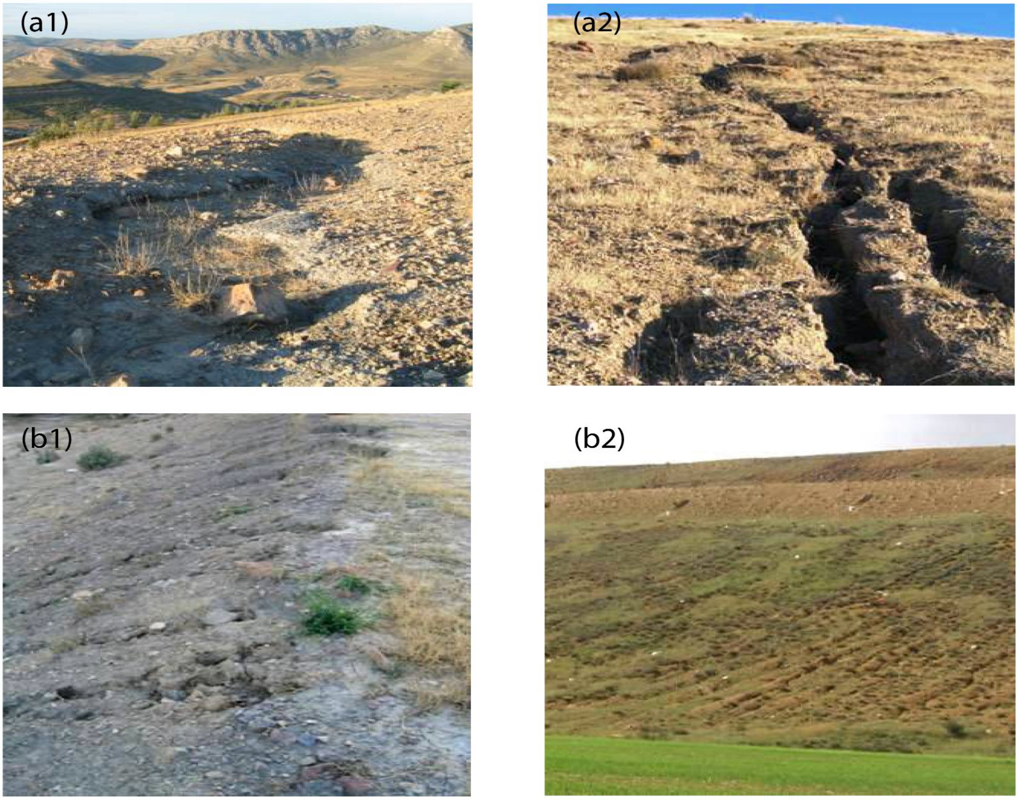

Imagen 1.4. Una parte muy considerable de los problemas de erosión hídrica superficial en laderas restauradas mineras están derivados de la presencia de errores en el diseño geomorfológico que favorecen la acumulación y concentración de volúmenes importantes de escorrentía. Las dos primeras imágenes (a1-a2) muestran las consecuencias que la ruptura y derivación de un canal de guarda en la parte alta de una ladera (a1) tiene en el desarrollo de procesos intensos de erosión en regueros (a2). Las otras dos imágenes (b1-b2) muestran el efecto producido por la presencia de áreas de contribución de escorrentía en la cabecera de dos laderas sobre el desarrollo de procesos de erosión con formación de regueros.

Tabla 1.1. Tasas de erosión en laderas restauradas derivadas de actividades mineras a cielo abierto.

Autores	Área de estudio	Método	Proceso predominante	t ha^{-1} año^{-1}
Clotet *et al.*, 1988	Cataluña España	Microcuencas 38 m^2	Erosión en regueros	140
Day, 1987	Nueva Gales del Sur Australia	Parcelas 2 m^2	Erosión laminar	1-10
Haigh, 1992	País de Gales Reino Unido	Agujas erosión	Erosión laminar Erosión en regueros	23 40
Kleeberg *et al.*, 2008	Lusacia Alemania	Microcuenca 33 m^2	Erosión en regueros	180
McKenzie y Studlick, 1979	Ohio EEUU	Trampas	Erosión laminar	1
Nicolau, 1996; 2002	Teruel España	Microcuencas 70-100 m^2	Erosión laminar Erosión en regueros	1-3 50-160
Porta *et al*., 1989	Cataluña España	Perfil regueros	Erosión en regueros	100-500

los procesos de erosión. Así, la vegetación actúa protegiendo el suelo frente al golpeteo de la lluvia y la tracción causada por los flujos superficiales de escorrentía (Elwell y Stocking, 1976; Francis y Thornes, 1990); del mismo modo, su desarrollo mejora las condiciones estructurales que regulan la capacidad de infiltración y erodibilidad del suelo a través del aporte de materia orgánica, dinámica radicular y estimulación de la actividad microbiana (Bochet *et al*., 1999; Durán-Zuazo y Rodríguez-Plequezuelo, 2008). En términos prácticos, estos efectos se han traducido en la identificación, como objetivo prioritario para la restauración de los espacios alterados, del establecimiento temprano de unos niveles mínimos de cubierta vegetal para el control de la erosión (Snelder y Bryan, 1995; Loch, 2000; Whisenant, 2005).

Efectivamente, el sistema suelo-vegetación está articulado mediante mecanismos de retroalimentación que regulan tanto el desarrollo de la vegetación y formación del suelo como el desarrollo de los procesos de erosión y sedimentación (Kirkby *et al*., 1998; Puigdefábregas *et al*., 1999). No obstante, gran parte de las investigaciones realizadas sobre estas relaciones se han centrado en el análisis unidireccional del efecto que la vegetación tiene sobre los procesos de erosión hídrica superficial. Este hecho demanda nuevos esfuerzos que integren el análisis de los impactos causados por la erosión sobre el desarrollo de la vegetación para una comprensión más completa de estas relaciones (Puigdefábregas, 1996; García-Fayos, 2004; Porder *et al*., 2005).

Efectos de la erosión sobre la vegetación en ambientes con restricciones hídricas

Diferentes trabajos realizados en ambientes naturales intensamente erosionados de clima mediterráneo-seco han evaluado el papel que la pérdida de semillas por fenómenos de erosión tiene sobre el establecimiento de la vegetación (García-Fayos *et al*., 1995; 1997; 2000). Los resultados obtenidos indican que estas pérdidas son en general pequeñas (<13%) y no explican la falta de vegetación que les caracteriza. El principal factor limitante para la colonización y el desarrollo vegetal sugerido en estos trabajos, así como en otros desarrollados en sistemas artificiales sometidos a procesos intensos de erosión (laderas de carreteras y minas) bajo el mismo marco climático, es la baja disponibilidad de agua en el suelo para las plantas (García-Fayos *et al*., 2000; Bochet y García-Fayos, 2004; Nicolau y Moreno-de las Heras, 2005). Así, la disponibilidad espacio-temporal del agua en el suelo es reconocida como uno de los principales factores de control para la estructura y la dinámica vegetal tanto en sistemas naturales como restaurados de clima mediterráneo-seco (Peco, 1989; Cantón *et al*., 2004; Martínez-Ruiz y Marrs, 2007; Tormo *et al*., 2008).

En este sentido, se puede señalar la existencia de un importante número de evidencias que apuntan al impacto que la erosión tiene sobre la disponibilidad de agua en el suelo como uno de los mecanismos fundamentales de interferencia para el desarrollo de la vegetación (Lal, 1998; Pimentel y Harvey, 1999). Los diferentes mecanismos por los que los procesos de erosión controlan la disponibilidad y distribución espacial de la humedad del suelo se manifiestan a diferentes escalas (Figura 1.5):

- A escalas pequeñas, de pedión, la pérdida de espesor de suelo producida por los procesos de erosión supone la reducción directa del almacén total de agua en el perfil del suelo (Graveel et al., 2002). Del mismo modo, a estas escalas pequeñas actuarían los procesos de debilitamiento y ruptura de los agregados del suelo, así como la formación de costras y sellos superficiales que, en conjunto, pueden reducir drásticamente la infiltración del agua en el suelo (Morin *et al*., 1989).

- A escalas mayores, de ladera, adquieren importancia los cambios micro-topográficos inducidos por la erosión, que tienen como consecuencia la alteración de los patrones de circulación y redistribución de los flujos de escorrentía a lo largo de las laderas. Así, es bien conocido que el agua que no se infiltra en un punto dado del espacio puede re-infiltrase ladera abajo (Kirkby, 2001). Este efecto de escala está fuertemente ligado a la presencia de variaciones espaciales de la capacidad de infiltración del suelo, almacenamiento superficial de la escorrentía (en pequeñas depresiones) y variaciones temporales en la precipitación y generación de escorrentía (Joel *et al*., 2002; Wainwright y Parsons, 2002; van de Giesen *et al*., 2005). La pérdida de rugosidad superficial en la dirección general de la pendiente (causada por el efecto abrasivo de la erosión) tiene como consecuencia directa la reducción de la capacidad de almacenamiento temporal y el aumento de la velocidad del flujo de escorrentía (Murphy y Flewin, 1993), reduciendo las posibilidades de re-infiltración ladera abajo. A este efecto se le suma el generado por la formación de redes integradas de regueros, que aumentan drásticamente la conectividad hidrológica en el espacio y proporcionan vías preferentes de circulación de los flujos (Favis-Mortlock *et al*., 2000; Bracken y Croke, 2007). En consecuencia, las posibilidades de re-infiltración de la escorrentía dentro de la ladera se ven drásticamente limitadas y el patrón espacial de distribución de la humedad del suelo se ve condicionado por la presencia de estas redes de drenaje superficial (Nicolau, 2002; Biemelt *et al*., 2005).

Estos mecanismos juegan un papel muy relevante para la comprensión de los procesos de degradación en ambientes con restricciones hídricas, como es el caso de los de clima mediterráneo seco, donde las interacciones entre los procesos hidrológicos y la vegetación son particularmente intensas. En estos ambientes la vegetación se distribuye en general de forma discontinua en el paisaje, definiendo un mosaico de áreas generadoras e importadoras de escorrentía y sedimentos entre las que los flujos superficiales son redistribuidos, limitando la pérdida de recursos y optimizando la producción primaria del sistema a escala de ladera (Puigdefábregas *et al*., 1999; Calvo-Cases *et al*., 2003; Wilcox y Newman, 2005). Diferentes tipos de perturbaciones (fuego, sobrepastoreo, sequía intensa, etc.) pueden provocar la alteración de la cubierta y distribución de la vegetación, produciendo la aceleración de los procesos de erosión y una pérdida neta de recursos por los mecanismos señalados previamente (Wilcox *et al*., 2003).

Algunos trabajos sugieren que una pérdida significativa de recursos hídricos causada por la

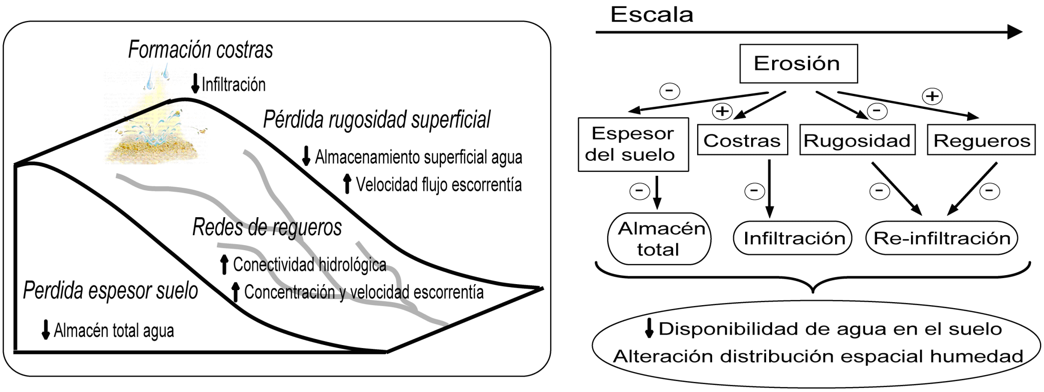

Figura 1.5. Diagrama conceptual que representa los diferentes mecanismos asociados a los procesos de erosión hídrica superficial que pueden condicionar la disponibilidad y distribución del agua en el suelo a diferentes escalas a lo largo de una ladera ideal.

aceleración de los procesos de erosión en estos ambientes puede inducir la activación de un proceso sostenido de degradación a largo plazo (Thornes, 1985; Whisenant, 2002; Wilcox *et al*., 2003; Sarah, 2005). Así, estas pérdidas de agua podrían afectar a la vegetación mediante el incremento de los niveles de estrés hídrico, reduciendo la producción de biomasa y de semillas e incluso comprometiendo las posibilidades de establecimiento de nuevos individuos y la persistencia de las especies con mayores requerimientos. La pérdida asociada de cubierta vegetal impulsaría nuevos incrementos en el desarrollo de los procesos de erosión y pérdidas de agua en forma de escorrentía superficial. A largo plazo, la concentración de los flujos de escorrentía y sedimentos y la formación asociada de redes de regueros y/o pequeñas cárcavas podría incluso afectar a la organización espacial de la vegetación, condicionando la distribución de la humedad del suelo y la perturbación mecánica (asociadas a los flujos concentrados) y, por tanto, el patrón espacial de supervivencia y crecimiento de la vegetación (Puigdefábregas y Sánchez, 1996; Saco *et al*., 2007).

Se han encontrado algunas evidencias que apoyan este patrón de interacción entre los procesos de erosión y la vegetación en ambientes mediterráneos intensamente erosionados. De esta forma, el desarrollo de procesos intensos de erosión se ha asociado en estos ambientes con la formación de procesos paralelos de degeneración en la vegetación, conduciendo a la formación de comunidades vegetales muy ralas, de baja productividad y muy poco diversas, en general caracterizadas por la presencia de unas pocas especies especialmente adaptadas a la escasez de agua (Guardia, 1995; Guerrero-Campo y Montserrat-Martí, 2004).

Interacciones erosión-vegetación y dinámica de sistemas restaurados mediterráneos

El conocimiento de los mecanismos de retroalimentación erosión-vegetación constituye una frontera conceptual en el marco de la estabilidad de los ecosistemas y presenta un notable interés aplicado de cara al éxito de las restauraciones (Davenport *et al*., 1998; Weltz *et al*., 1998; Osterkamp y Joseph, 2000). En este sentido, uno de los esfuerzos más originales en el marco ambiental mediterráneo para la comprensión de estos mecanismos y sus repercusiones en la evolución de los ecosistemas está representado por el modelo teórico de interacción dinámica entre el desarrollo de la vegetación y la erosión de John B. Thornes, en el que ambos procesos actúan como especies que compiten por la humedad del suelo como recurso limitante (Thornes, 1985; 1990; 2004).

En este modelo dinámico (Cuadro 1.3) ambos procesos (el desarrollo vegetal y el desarrollo de los procesos de erosión) interactúan entre sí limitando el desarrollo del otro. En este sentido la vegetación impulsa el desarrollo edáfico, modificando las características del suelo (aumentando la capacidad de infiltración y la erodibilidad del suelo) y en consecuencia limitando el desarrollo de los procesos de erosión. Asimismo, el desarrollo de los procesos de erosión limita el desarrollo de la vegetación mediante la reducción de la disponibilidad de agua para las plantas y la pérdida de nutrientes. La resolución de este modelo teórico prevee, en función de la situación de partida (cubierta vegetal y tasas de erosión) y las características del sistema de ladera (pendiente, longitud, tipo de suelo, régimen climático, etc.), dos tipos principales de evolución posible:

- Uno marcado por el dominio de los mecanismos desencadenados por el desarrollo vegetal sobre la dinámica del ecosistema, que conduciría a éste hacia estados de máximo desarrollo (máxima cubierta vegetal y tasas de erosión mínimas).

- Otro marcado por el dominio de los procesos de erosión en la dinámica temporal del ecosistema, que conduciría a éste hacia estados de máxima degradación (tasas de erosión máximas y desarrollo de la cubierta vegetal mínimo).

De forma análoga a los resultados predichos por este modelo dinámico teórico, se pueden identificar dos grandes patrones de evolución en los sistemas restaurados mineros (Nicolau y Asensio, 2000):

- Uno representado por la prevalencia del control biológico sobre las respuestas hidrológicas, en el que la vegetación participa activamente modificando las características del sistema. Como aspectos característicos de este patrón de evolución se pueden señalar la mejora de la capacidad de infiltración y la disminución de la erodibilidad del suelo a lo largo del tiempo, así como una reducción progresiva de los volúmenes de escorrentía generados (en general como flujos laminares) y de la erosión, que se asocian al desarrollo progresivo de la cubierta vegetal (Sanchez y Wood, 1986; Nicolau, 1996; Loch, 2000).

- Otro representado por la prevalencia del control abiótico, en el que fenómenos asociados a los procesos de erosión (formación de costras superficiales y de redes de regueros, desmantelamiento de la rugosidad superficial en la dirección general de la pendiente, etc.)

Cuadro 1.3. Modelo dinámico de interacción erosión-vegetación de John B. Thornes

Este modelo parte de la premisa de que en un ambiente dado (caracterizado por unos niveles particulares de precipitación y unas características topográficas y edáficas particulares) la erosión y el desarrollo de la vegetación, cuando son analizados independientemente (en ausencia del otro proceso), crecen a lo largo del tiempo siguiendo una patrón logístico (de forma análoga a como lo haría una especie en ausencia de competencia interespecífica), hasta llegar a unos niveles máximos de desarrollo (Thornes, 1985). En el caso de la erosión, este patrón de crecimiento se explica por la pérdida asociada de la capacidad de infiltración del suelo (por formación de costras superficiales y emergencia de niveles del suelo con mayor densidad) que aumentaría el volumen de escorrentía superficial, reforzando el proceso de erosión a lo largo del tiempo hasta alcanzar unos niveles máximos determinados por la precipitación y los fenómenos de agotamiento del material erosionable. En el caso de la vegetación, el patrón de crecimiento estaría explicado por la mejora de las condiciones del suelo (capacidad de infiltración), que la permitirían maximizar el uso del agua para la producción vegetal hasta alcanzar los niveles máximos determinados por la precipitación y la disponibilidad de otros recursos limitantes, como es el caso de los nutrientes.

Estos patrones de crecimiento se ven condicionados al hacer interactuar entre sí ambos procesos. Así, la vegetación inhibe el desarrollo de la erosión debido al aumento de las tasas de infiltración y la reducción de la erodibilidad del suelo. Este efecto fue introducido en el modelo mediante un término en la función de crecimiento de la erosión que refleja una reducción en las tasas de erosión de carácter exponencial ligada al desarrollo de la vegetación (Thornes, 1990). Este término está basado en la forma clásica de la relación que explica el impacto de la cubierta vegetal sobre el desarrollo de los procesos de erosión a través de una función

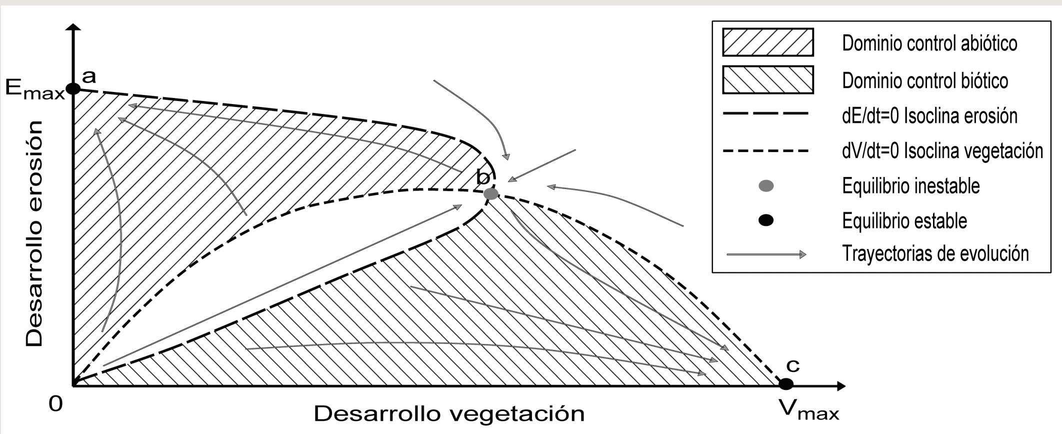

Figura 1.6. Diagrama que representa el efecto combinado del desarrollo de la vegetación y de los procesos de erosión en la evolución temporal de los sistemas de ladera en ambiente mediterráneo predicho por el modelo dinámico de John B. Thornes. Se pueden distinguir dos áreas en las que, a partir de diferentes situaciones iniciales, los mecanismos de retroalimentación de los procesos de erosión (dominio abiótico) o bien de la vegetación (dominio biótico) dirigen la evolución del sistema hacia estados estables de máxima degradación (a) o en su caso máximo desarrollo vegetal (c); el resto de trayectorias posibles conducen a un punto de equilibrio inestable (b), a partir del cual el sistema tenderá a dirigirse hacia las situaciones estables a ó c en función de factores no deterministas (fluctuaciones climáticas, perturbaciones, etc.). Tomado con modificaciones de Thornes (2004).

exponencial negativa (Elwell y Stocking, 1976; Francis y Thornes, 1990; Durán-Zuazo y Rodríguez-Plequezuelo, 2008). Por otro lado, la erosión inhibe el desarrollo de la vegetación mediante la pérdida de recursos hídricos y fertilidad del suelo. Este efecto fue introducido inicialmente en el modelo mediante un término en la función de crecimiento de la vegetación que refleja una caída lineal de la producción vegetal ligada a las tasas de erosión (Thornes, 1985). No obstante, en trabajos posteriores se señaló la necesidad de modificar este término para que esta relación fuese reflejada en base a una relación exponencial negativa, más acorde a la realidad del fenómeno (Thornes, 1990; Thornes y Brandt, 1994).

Las dos funciones de crecimiento (la de la vegetación y la de la erosión, con sus términos de interacción incluidos) definen un sistema dinámico característico para unas condiciones de precipitación, edáficas y topográficas dadas (Figura 1.6). En este se pueden identificar diferentes regiones críticas para valores de vegetación y erosión que marcan la evolución posible del sistema. De este modo, dadas unas condiciones de partida (cubierta de vegetación y tasas de erosión) el sistema podría evolucionar siguiendo trayectorias dominadas en unos casos por mecanismos de control abiótico (área 0-a-b; Figura 1.6) o en otros casos por mecanismos de control bióticos (área 0-c-b; Figura 1.6). En el primero de los casos (dominio abiótico), los mecanismos de retroalimentación gobernados por los procesos de erosión conducirían al sistema hacia estados de máxima degradación (tasas de erosión máximas y desarrollo de vegetación mínimo). En el segundo caso (dominio biótico), los mecanismos de retroalimentación gobernados por la vegetación conducirían al sistema hacia estados de máximo desarrollo vegetal, reduciendo las tasas de erosión a niveles mínimos.

gobiernan la hidrología del sistema. Como aspectos característicos de este modo de evolución se pueden señalar la organización del sistema acorde a un patrón de unidades de regueros e inter-regueros, el deterioro progresivo de los niveles de cubierta vegetal introducidos en las operaciones de revegetación (ya de por sí bajos en su inicio) y el mantenimiento de las tasas de erosión en niveles altos o en su caso, un aumento de las mismas sostenido en el tiempo (Haigh, 1980; Nicolau, 1996; 2002).

Este conjunto de consideraciones permiten intuir la existencia de unos umbrales críticos de equilibrio entre la vegetación y la erosión en torno a los cuales (por encima y por debajo), mecanismos de retroalimentación de diferente naturaleza pueden impulsar a los sistemas restaurados hacia resultados muy distintos. Si bien es cierto que estos umbrales de estabilidad difieren en función del sistema con el que se trabaje (García-Fayos, 2004), parecen existir algunas manifestaciones de carácter general que pueden contribuir a la interpretación e identificación de los mismos. Así, en sistemas de ladera de ambientes secos, el desencadenamiento de procesos sostenidos de degradación de la vegetación parece estar íntimamente vinculado al desarrollo de redes integradas de drenaje superficial (constituidas por regueros o pequeñas cárcavas), que incrementan drásticamente la conectividad estructural del sistema (Turnbull *et al*., 2008). Cabe añadir que la formación de regueros por erosión hídrica superficial en laderas restauradas mineras es un fenómeno muy frecuente, especialmente en sus fases iniciales (Riley, 1995). En este sentido, trabajos previos realizados en laderas restauradas mineras en condiciones climáticas mediterráneo-continentales de carácter seco advierten que cuando los niveles iniciales obtenidos de desarrollo vegetal y erosión del suelo no posibilitan la disipación de los procesos de formación de redes de regueros (debido fundamentalmente a errores en el diseño geomorfológico y/o errores en el manejo de los sustratos) el desarrollo de la vegetación se puede ver condicionado a largo plazo, posiblemente a causa de la pérdida de recursos hídricos (Nicolau, 2002; 2003). Así, el análisis de los procesos de interacción entre los fenómenos de erosión en regueros y desarrollo vegetal puede contribuir de forma significativa a la comprensión de los mecanismos subyacentes a los efectos ecológicos de la erosión en estos ambientes artificiales y a la mejora de la práctica de las restauraciones en el marco climático mediterráneo.

Objetivos, hipótesis y escala de trabajo

La presente tesis se puso en marcha en el seno de los proyectos PI2004/024 (Universidad de Alcalá) y CGL2004-00355/BOS (CICYT), desarrollados en el marco de la restauración ecológica de áreas degradadas en ambientes mediterráneos-continentales. Ambos proyectos fueron diseñados en base a la identificación y análisis de los procesos que regulan la disponibilidad y el uso del agua en ambientes restaurados con restricciones hídricas; para ello, sus objetivos pusieron una atención especial en el estudio de las interacciones entre el desarrollo de la vegetación y los procesos de erosión hídrica superficial.

El objetivo general de esta tesis doctoral es determinar los efectos de la erosión sobre la disponibilidad y la distribución del agua en el suelo, y sus consecuencias sobre la dinámica de las comunidades vegetales y el desarrollo de los procesos biológicos del suelo en ecosistemas restaurados de ladera derivados de la minería a cielo abierto del carbón en ambiente mediterráneo-continental.

La hipótesis de partida es que entre los procesos más influyentes en la evolución de las laderas restauradas se encuentra la erosión en regueros, que intensifica el déficit hídrico, primer factor limitante del éxito de las restauraciones mineras en clima mediterráneo-continental.

Los objetivos específicos planteados son:

1. Identificar las principales trayectorias de evolución y fuerzas directoras de la sucesión vegetal en laderas restauradas mineras de ambiente mediterráneo-continental, determinando el papel representado por los procesos de erosión hídrica superficial en el marco de la dinámica temporal de estos ecosistemas artificiales.

2. Determinar los mecanismos de regulación de la generación de escorrentía y sedimentos que operan en los suelos mineros, atendiendo al impacto que procesos de control abiótico (desarrollo de costras superficiales) y biótico (desarrollo de la vegetación) puedan tener sobre su dinámica estacional.

3. Determinar el impacto de los procesos de erosión hídrica superficial en el patrón de circulación de los flujos de escorrentía y sedimentos, con especial atención en la redistribución espacial de los recursos (agua y sedimentos) a lo largo de las laderas.

4. Determinar el impacto de los procesos de erosión hídrica superficial en la disponibilidad y distribución espacial de la humedad del suelo.

5. Caracterizar la respuesta de la vegetación al déficit hídrico y a otras formas de explotación derivadas de la erosión a través de su impacto sobre el estado hídrico y estructura de la vegetación, y sus efectos sobre los procesos de colonización vegetal.

6. Determinar el impacto de los procesos de erosión hídrica superficial en el desarrollo de la estructura física y funciones biológicas de los suelos mineros.

7. Analizar la existencia de umbrales críticos (en términos de cubierta vegetal y tasas de erosión) que puedan definir trayectorias de evolución diferentes (dominadas por mecanismos de control biótico o en su caso abióticos) para los sistemas de ladera estudiados.

El desarrollo de los objetivos propuestos en esta tesis ha incluido una aproximación a dos escalas de trabajo diferentes. Una regional, llevada a cabo en la "cuenca lignitífera de Teruel", enfocada a la identificación de este problema en laderas restauradas mineras en un entorno territorial representativo de las condiciones climáticas mediterráneas-continentales. Otra local, enfocada al análisis experimental de los efectos ecológicos particulares causados por los procesos de erosión hídrica superficial en estos ambientes artificiales de ladera. Esta última se ha desarrollado a partir del análisis de la dinámica hidrológica, vegetal y edáfica en un conjunto formado por cinco laderas mineras ("Área experimental Utrillas") en las que el desencadenamiento temprano de procesos de erosión de diferente intensidad ha determinado, desde su construcción en 1988-89, su evolución temporal.

Estructura de la memoria

Además del presente capítulo introductorio (**capitulo 1**), esta memoria doctoral esta constituida por otros ocho capítulos.

El **capítulo 2** comprende la descripción del área de estudio donde se han llevado a cabo los trabajos que integran esta tesis. En este capítulo se puede encontrar una descripción general de la geografía, clima, conservación y usos del territorio, así como de la dimensión ambiental de la minería a cielo abierto del carbón en la "cuenca lignitífera de Teruel". Del mismo modo, se puede encontrar una descripción pormenorizada de las cinco laderas que integran el "Área experimental Utrillas".

Los seis capítulos que se encuentran a continuación (**capítulos 3 a 8**) recogen los resultados de los diferentes trabajos realizados para el desarrollo de los objetivos propuestos. Estos capítulos se corresponden con artículos científicos ya publicados o enviados a revistas internacionales especializadas. Para los mismos se ha mantenido la estructura fiel a su versión final (incluido el idioma -inglés-), aún a sabiendas de que este tipo de organización puede generar cierta redundancia en los contenidos, especialmente en las secciones relacionadas con la descripción del área de estudio y los métodos empleados.

El **capítulo 3** desarrolla el objetivo específico 1. Este trabajo (realizado a escala regional en la "cuenca lignitífera de Teruel") se centra en la identificación de los grandes patrones de sucesión vegetal en laderas restauradas mineras del territorio. Para abordarlo se analizó un bloque de información extensivo obtenido en laderas de diferentes minas de la región durante dos campañas de muestreo: una desarrollada en 1987-88 (datos no publicados obtenidos por el Dr. José Manuel Nicolau) y otra desarrollada en 2002-03.

El **capítulo 4** desarrolla el objetivo específico 2. Este trabajo (llevado a cabo en el "Área experimental Utrillas") se centra en el análisis de la respuesta hidrológica (generación de escorrentía y sedimentos) y la dinámica estacional de los suelos mineros, con especial atención al impacto que el desarrollo de la vegetación y los procesos de formación de costras superficiales tienen a pequeña escala sobre estas respuestas. Fue abordado mediante experimentación con lluvia simulada y se responde a una cuestión de interés general a todos los ambientes ecológicos: ¿Cómo es el control de la vegetación sobre los procesos erosivos?.

El **capítulo 5** desarrolla el objetivo específico 3. Este trabajo (llevado a cabo en el "Área experimental Utrillas") estudia el impacto que los procesos de erosión con formación de redes de regueros tienen en las variaciones que se producen al cambiar de escala (longitud) en la generación de escorrentía y sedimentos, alterando la redistribución de recursos dentro de las laderas. Este trabajo fue abordado a partir de la monitorización en campo de la formación de escorrentía y sedimentos en parcelas de diferente longitud durante un año hidrológico (2005-06).

El **capítulo 6** desarrolla el objetivo específico 4 y parcialmente el objetivo específico 5. Este trabajo (llevado a cabo en el "Área experimental Utrillas") estudia el impacto que los procesos de erosión con formación de redes de regueros tienen en la distribución y disponibilidad de agua en el suelo para las plantas y sus efectos sobre la vegetación. Para abordar este trabajo se monitorizó durante un año hidrológico (2005-06) la humedad edáfica en una red de sensores distribuidos a diferentes profundidades del suelo. Asimismo se analizó la estructura de la vegetación (biomasa y riqueza específica) así como el estado hídrico, distribución espacial y germinabilidad potencial en condiciones de campo de la especie mayoritaria presente en las laderas experimentales.

El **capítulo 7** concluye el desarrollo del objetivo específico 5. Este trabajo (llevado a cabo en tres de las cinco laderas que componen el "Área experimental Utrillas") estudia el impacto que los procesos de erosión con formación de redes de regueros tiene en los procesos de colonización y establecimiento vegetal. Para abordar este trabajo, durante un ciclo de crecimiento vegetal (2003-04), se analizó el banco de semillas del suelo (densidad y riqueza) así como la emergencia (tasas) y mortalidad (tasas y causas) de plántulas en campo, y la producción final de semillas (densidad).

El **capítulo 8** desarrolla el objetivo específico 6. Este trabajo (llevado a cabo en el "Área experimental Utrillas") estudia el impacto que los procesos de erosión tienen en el desarrollo de la estructura física y funcionalidad biológica de los suelos mineros. Para abordarlo se analizó la estructura física del suelo (distribución de tamaños y estabilidad de los agregados del suelo), así como una serie de parámetros descriptores de las poblaciones microbianas del suelo (tamaño y actividad) y de las rutas fundamentales de reciclado de los elementos (actividad de algunas enzimas hidrolíticas específicas de las rutas del fósforo, carbono y nitrógeno).

A continuación de estos seis capítulos que recogen los resultados experimentales, se presenta un nuevo capítulo (**capítulo 9**) que constituye la síntesis de los resultados más destacados, cuya integración conjunta permite abordar el último objetivo propuesto (objetivo específico 7), proponiendo una serie de umbrales prácticos (en términos de cubierta de vegetación y tasas de erosión) para la diferenciación de trayectorias de evolución aplicables a los sistemas de ladera artificiales estudiados. Junto a la síntesis general de los resultados, en este mismo capítulo se pueden encontrar las conclusiones principales de esta tesis y las recomendaciones derivadas para la práctica de restauraciones en sistemas artificiales de ladera de clima mediterráneo.

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Capítulo 2

Capítulo 2

Área de estudio

La cuenca lignitífera de Teruel

Geografía y clima

La cuenca lignitífera de la provincia de Teruel queda delimitada por las coordenadas 40º 45' y 41º 00' de latitud Norte y 0º 15' y 1º 00' de longitud Oeste. Territorialmente se encuentra situada en la vertiente meridional del valle del Ebro (rama suroriental del Sistema Ibérico) entre las cuencas de los ríos Martín y Guadalope (Figura 2.1).

El clima de la región es Mediterráneo templado de carácter seco (*sensu* Papadakis, 1966) con pequeñas variaciones asociadas a la altitud, que varía entre los 1100 y 700 m sobre el nivel del mar. Este régimen climático está caracterizado por unas temperaturas medias anuales comprendidas entre los 11 y 14ºC, volúmenes anuales medios de precipitación comprendidos entre 400 y 500 mm, y valores asociados de evapotranspiración potencial (*sensu* Thornthwaite, 1948) comprendidos entre 650 y 750 mm (Peña-Monné *et al.*, 2002). A lo largo del territorio en dirección Este-Oeste se puede identificar un gradiente de continentalidad que diferencia dos áreas bien definidas: el clima de los dos tercios occidentales de la región (triángulo comprendido entre los municipios de Portalrubio, Aliaga, y Estercuel, que se corresponden con los tramos alto y medio del río Martín y alto del río Guadalope) presenta un marcado carácter continental, con amplitudes térmicas anuales (diferencia entre las temperaturas máximas y mínimas absolutas) en torno a 45-50ºC y un periodo de heladas muy prolongado, generalmente desde principios de octubre hasta el final de abril. En el tercio oriental (triángulo formado por los municipios de Berge, Castellote y Calanda, que se corresponde con el tramo medio del río Guadalope) el rigor continental se ve ligeramente atenuado por una mayor influencia marítima mediterránea, presentando amplitudes térmicas en torno a 43-46ºC y un periodo de heladas menos prolongado, generalmente desde principios de noviembre al inicio de abril (de León-Llamazares, 1991).

El régimen regional de precipitaciones está caracterizado por la concentración de los episodios lluviosos en las estaciones de primavera y otoño, y por un período de sequía estival en torno a tres meses entre junio y septiembre, en el que la evapotranspiración supera las aportaciones de agua en forma de precipitaciones (250-300 mm de déficit hídrico). La tónica general del régimen de precipitaciones está marcada por la irregularidad interanual, con un promedio de 50 a 60 precipitaciones al año. Por su alta intensidad, cabe destacar entre éstas las tormentas convectivas estivales así como las precipitaciones torrenciales otoñales asociadas a fenómenos de gota fría e inestabilidad mediterránea, que en ambos casos pueden llegar a alcanzar intensidades máximas de precipitación superiores a los 100 mm

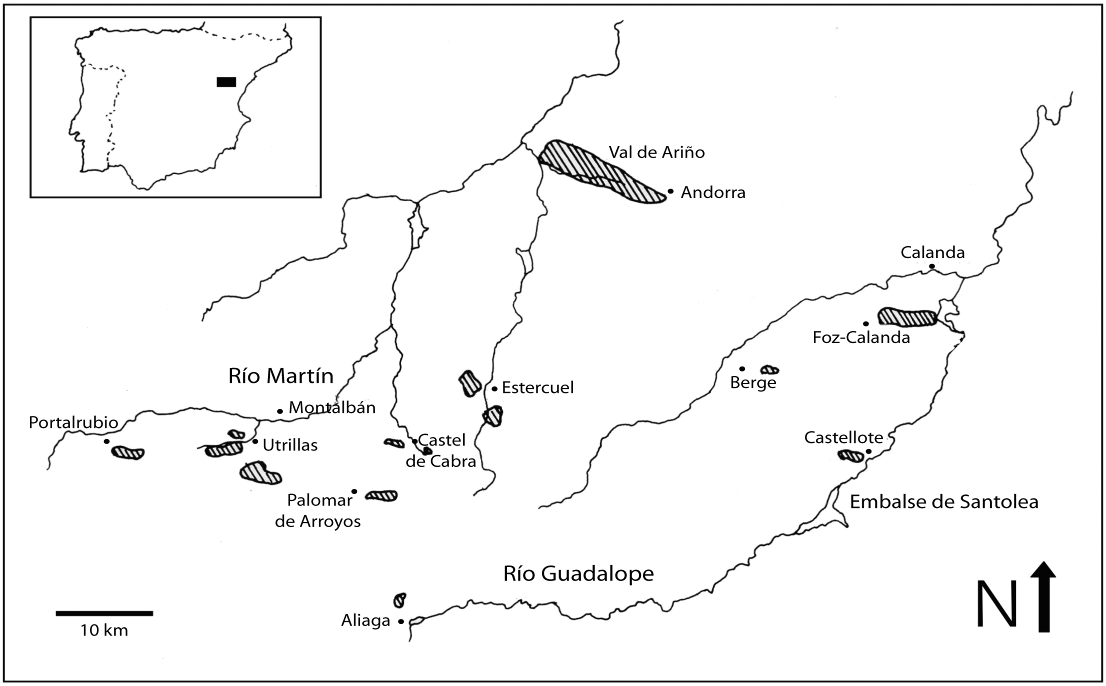

Figura 2.1. Mapa de localización de la cuenca lignitífera de la provincia de Teruel. Las áreas rayadas representan la superficie ocupada por las explotaciones a cielo abierto.

en 24 horas (Peña-Monné *et al*., 2002).

El conjunto de las condiciones climáticas de la zona (intensa sequía estival y periodo prolongado de heladas) plantea limitaciones importantes para el crecimiento de la vegetación, que en general se desarrolla en el espacio de forma discontinúa. Este hecho, unido a la torrencialidad de las precipitaciones y en ocasiones a la abrupta geomorfología del paisaje hacen de este entorno una región especialmente vulnerable a la formación de procesos de erosión intensos, como puede ser evidenciado por la presencia frecuente de pequeños sistemas de cárcavas en el territorio. Este fenómeno es común a las áreas comprendidas en la vertiente mediterránea española, donde la agresividad climática unida a factores litológicos, topográficos y al uso ancestral del territorio hacen que, en general, el riesgo de erosión sea considerablemente alto (López-Bermúdez, 1990; Cerdà, 2001).

Usos y estado de conservación de la vegetación y los suelos

La vegetación potencial de la región varía desde la dominancia de *Quercus ilex* a la de *Q. faginea* en las cotas más elevadas (Rivas-Martínez, 1987). La utilización humana del territorio ha sido muy intensa desde antiguo. Así lo atestiguan los abundantes asentamientos celtibéricos hallados y el denso poblamiento establecido en los siglos XII y XIII, tras la ocupación cristiana de la región. Los principales usos que históricamente han afectado al territorio han sido la ganadería, la agricultura de secano y la extracción de leña.

En paralelo al gradiente climático de continentalidad anteriormente descrito se puede identificar una importante variación en el grado de conservación de la vegetación. Así, la vegetación presente en el tercio oriental de la región (extremo de menor continentalidad) está compuesta por un mosaico configurado por numerosas manchas forestales bien conservadas (dominadas por *Q. ilex* y *Pinus halepensis*), algunos cultivos de secano (cereal y almendro principalmente) y terrazas de cultivo abandonadas, colonizadas por diferentes especies de matorral (*Rosmarinus officinalis*, *Genista scorpius*, *Pistacea lentiscus*, *Thymus vulgaris*, *Artemisia campestris* y *Dorycnium pentaphyllum*). En los dos tercios occidentales (extremo de mayor continentalidad) la vegetación presente está más degradada, probablemente debido a condiciones ambientales más limitantes para su recuperación. Las áreas forestales son escasas y la mayor parte del paisaje está cubierto por manchas muy discontinuas de matorral (dominadas por *G. scorpius* y *T. vulgaris*) y cultivos de cereal (Montserrat, 1990).

En general, los suelos de la región varían entre *Typic* y *Lithic Xerorthent* a *Calcic Xerochrept* (*sensu* Soil Survey Staff, 1998), con un pH neutro o moderadamente básico (7.3-8.6), pobres en materia orgánica (0.5-2.1%), texturas franco-arcillosas o franco-arcillo-arenosas y pedregosidad comprendida entre el 20 y 47% en peso (Arranz, 2004).

Reservas de carbón y caracterización de la actividad minera

Las reservas regionales de carbón, constituidas por 908 Mt de hulla subitominosa (lignito negro), representan aproximadamente el 18% en peso bruto (12% en unidades equivalentes de energía) del total de las reservas de carbón de la nación (IGME, 1985). La génesis de los carbones turolenses se remonta a hace unos 120 millones de años, a partir del enterramiento de restos vegetales en ambientes deltaicos, marismas y medios fluviales. Los yacimientos de carbón se encuentran en dos formaciones geológicas consecutivas (formación "Escucha" y formación "Utrillas"), cretácicas de edad Albiense (Gutiérrez-Elorza, 1985), aunque los niveles de carbón explotables se concentran principalmente en la primera de las formaciones (Albéniz-Campás, 1993). El conjunto de materiales estériles que acompañan a los niveles de carbón están constituidos por arenas, conglomerados y arcillas con caolín, procedentes de la formación "Escucha" (serie originada por la retirada del mar), así como areniscas, arenas y arcillas versicolores, procedentes de la formación "Utrillas" (originada en un medio fluvial anastomosado). Los materiales estériles se completan con conglomerados, areniscas, arenas, arcillas, calizas, margas, yesos, materiales aluviales y glacis de origen terciario y cuaternario (IGME, 1979). Estos materiales estériles tienen unas características fisicoquímicas muy heterogéneas, abarcando un gradiente desde valores de pH similares a los de los suelos del entorno natural hasta valores notablemente ácidos (en torno a 3-4), asociados con frecuencia a la presencia de materiales piríticos (Arranz, 2004).

Los análisis realizados por el arqueólogo Javier Ibáñez González (comunicación personal) localizan los primeros vestigios de actividad minera en la cuenca lignitífera de Teruel en la Edad del

Hierro, con la formación de pequeñas minas y hornos de fundición, cuya actividad se prolongó cerca de 300 años. Posteriormente, asentamientos cartaginenses en la Edad Antigua recuperaron la actividad, que no cesó hasta entrada la Edad Media (Miana-Escabosa y Valero-Ruiz, 2003). En el siglo pasado se inició la explotación a gran escala de los lignitos turolenses, mediante explotaciones subterráneas. En 1900 se constituye la empresa Minas y Ferrocarril de Utrillas (MFU) en el municipio que la da nombre; posteriormente en 1914 comienza la explotación en la Val de Ariño. La trayectoria histórica de la minería del carbón en Teruel se ha caracterizado desde entonces por su irregularidad. Entre las causas de esta dinámica se encuentran la dependencia de mercados externos, la falta de iniciativa privada y pública en la modernización, y la escasa calidad del carbón explotado.

La construcción, a finales de la década de los años 70 de la Central Térmica "Teruel" en Andorra generó un nuevo impulso en la extracción, sustentado en el desarrollo de la minería a cielo abierto (Nicolau y Ruiz, 1986). Así, en 1976 la empresa Sociedad Minera Catalana Aragonesa (SAMCA) inició la explotación industrial de carbón a cielo abierto en la Val de Ariño. En 1980 se sumaron los primeros desmontes a cielo abierto por parte de la Compañía General Minera de Teruel en Estercuel y en Utrillas por parte de MFU, así como la apertura, en 1981, de la Corta Alloza por parte de ENDESA (Abril y Molina, 1997). El ritmo acelerado de apertura de nuevas minas se mantuvo hasta mediados de la década de los años ochenta, con la incorporación de numerosas pequeñas empresas mineras locales.

La baja calidad de los lignitos turolenses y el menor precio de los carbones extranjeros redujeron a partir de 1984 la demanda regional de carbón, desembocando en una restructuración del sector. Numerosas minas de pequeño y mediano tamaño cerraron durante la segunda mitad de la década de los años ochenta. A este fenómeno se le sumó la clausura progresiva de las explotaciones subterráneas. Desde entonces, el ritmo de apertura de nuevas minas a cielo abierto ha sido mucho más bajo. Tras el fin de explotación en el año 2001 de la compañía MFU, en la actualidad la explotación a cielo abierto del carbón es llevada a cabo fundamentalmente por dos grandes compañías de dimensión nacional: SAMCA y ENDESA (Miana-Escabosa y Valero-Ruiz, 2003).

Dimensión ambiental de la minería a cielo abierto de carbón

Desde 1976 y a lo largo de las tres décadas de explotación a cielo abierto, se han abierto un total de 24 minas de carbón, ocupando una superficie aproximada de 3000 hectáreas (el 0,20% de la superficie de la provincia de Teruel), de las que 800 se encuentran a día de hoy en explotación activa (Mellado-García, 2006). El legado ambiental de estos 30 años de actividad puede resumirse señalando que el 67% de la superficie alterada (descontando la que se encuentra actualmente en explotación) ha sido restaurada con éxito (plataformas agrícolamente productivas y laderas sin regueros, con buen desarrollo de la vegetación), mientras que el tercio restante constituye el pasivo ambiental de esta actividad. Este pasivo está constituido por explotaciones no restauradas (aprox. 450 ha) y restauradas sin éxito (aprox. 280 ha). El principal problema ambiental que plantean estas superficies improductivas es de carácter hidrológico, afectando a los cauces naturales por la emisión de escorrentía y sedimentos (Nyssen, 2007).

Estas diferentes situaciones que componen el legado ambiental de la minería a cielo abierto del carbón en la región están asociadas con la evolución temporal de la legislación aplicable (introducción por ley de la obligación a restaurar los espacios alterados) y la mejora de las técnicas de restauración minera. Así, los espacios no restaurados corresponden a explotaciones de pequeño y mediano tamaño que comenzaron a operar antes de la aprobación del RD 2994/1982 de 15 de octubre, que introducía la restauración de las áreas alteradas como obligación legal para las actividades mineras. Entre las áreas alteradas sobre las que se acometieron operaciones de restauración se pueden diferenciar dos grandes tipologías, atendiendo a las técnicas y diseño empleado (Mellado-García, 2006): las realizadas durante la década de los años 80 (restauraciones de primera generación) y las realizadas a partir de los años 90 (restauraciones de segunda generación).

La principal diferencia entre las restauraciones de primera y segunda generación se basa en el diseño geomorfológico. Las restauraciones de primera generación (Imagen 2.1a) aplicaron formas abruptas a la modelación de las escombreras, basando su diseño en la creación de plataformas extensas, taludes de elevada pendiente (20-35º) y sistemas para la regulación de los flujos superficiales poco elaborados, organizados mediante cunetas. Este modelo de restauración, en numerosos casos, se caracteriza por el escaso éxito obtenido en el establecimiento de la vegetación y el desarrollo de procesos intensos de erosión hídrica superficial (Nyssen y Nicolau, 2008). Las restauraciones de segunda generación (Imagen 2.1b) están caracterizadas por el uso de formas topográficas más suaves, reduciendo la pendiente general de las laderas (18-22º), y el desarrollo de sistemas de regulación de los flujos superficiales más elaborados (cunetas, balsas de sedimentación, estructuras de seguridad, etc.). Dentro de la tipología de restauraciones de segunda generación, el sistema más original que se ha aplicado es el basado en

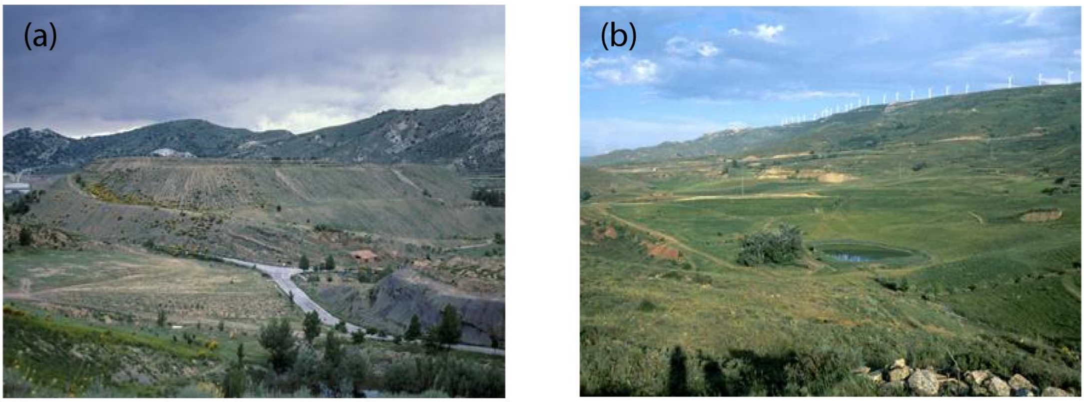

Imagen 2.1. Modelos de restauraciones mineras presentes en la cuenca lignitífera de la provincia de Teruel: (a) restauración de primera generación (mina El Murciélago, Utrillas); (b) restauración de segunda generación basada en cuencas compartimentadas (mina Alemanes, Utrillas).

cuencas compartimentadas conectadas mediante redes de drenaje naturalizadas (Nicolau, 2003); este modelo de restauración ha sido aplicado con éxito por la empresa MFU en la cuenca de Utrillas y se encuentra actualmente en experimentación en las cuencas de Estercuel-Gargallo por parte de la empresa minera ENDESA (Pérez-Domingo *et al.*, 2008).

Los trabajos de investigación realizados en la presente tesis se han llevado a cabo principalmente en laderas mineras procedentes de ambientes restaurados de primera generación y situaciones de transición hacia modelos de segunda generación.

Área experimental Utrillas

El "área experimental Utrillas" se encuentra ubicada en la mina restaurada El Moral, en el término municipal de Utrillas. Geográficamente se sitúa en el tercio más occidental de la cuenca lignitífera de la provincia de Teruel (40º47'24'' N, 0º47'24'' O) sobre la vertiente norte de la Sierra de San Just, a 1100 m de altitud.

De acuerdo con la clasificación de Papadakis (1996), el clima es mediterráneo templado y el régimen de humedad es seco. El régimen térmico local está marcado por un fuerte carácter continental. La temperatura media anual es 11ºC (7ºC en diciembre y 24ºC en julio), con un largo periodo de heladas que se extiende desde el inicio del mes de octubre hasta el final de abril. La precipitación media anual es 466 mm, principalmente concentrada en primavera y otoño, y la evapotranspiración potencial (*sensu* Thornthwaite, 1949) es 758 mm. Por tanto, el déficit hídrico asciende a un volumen de 292 mm, concentrado entre los meses de junio y octubre (Nicolau, 2002).

La mina El Moral toma su nombre del cerro en el que se ubica. Esta mina fue explotada a lo largo de la década de los años 80 por la empresa minera MFU. El área experimental está compuesta por cinco laderas mineras restauradas que se ubican de forma adyacente sobre la escombrera en la cara norte del cerro (Imagen 2.2).

Estas cinco laderas fueron restauradas a lo largo de los años 1988 y 1989, aplicando tratamientos de restauración muy similares. Las laderas presentan una pendiente general de 20º; el sustrato que se empleó para cubrir la superficie de las mismas (aproximadamente 1 m de espesor) es un material estéril procedente de la formación geológica "Escucha", de edad Albiense. Este sustrato tiene una textura franco-arcillosa (arcillas de mineralogía caolinítica-ilítica) y pH básico (Tabla 2.1). Atendiendo a la terminología de Rengasamy *et al*. (1984), se clasifica como no salino y no sódico. La preparación del suelo para las operaciones de revegetación consistieron en el labrado transversal a la pendiente, lo que per-

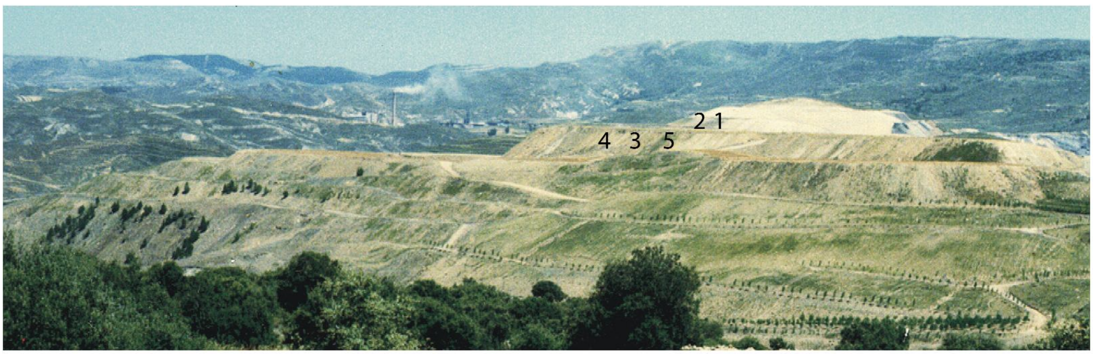

Imagen 2.2. Ubicación de las cinco laderas que componen el Área experimental Utrillas en la escombrera de la mina El Moral (cara norte del cerro).

Tabla 2.1. Características físico-químicas generales del sustrato empleado para la restauración de las laderas (media ± desviación).

	N	Ladera 1	Ladera 2	Ladera 3	Ladera 4	Ladera 5
Pedregosidad (%)	25	22,2 ± 2,2	24,7 ± 3,5	26,2 ± 4,1	25,2 ± 2,6	24,5 ± 3,3
Arena (%)	25	33,6 ± 3,6	33,5 ± 3,7	33,8 ± 3,0	39,9 ± 1,8	36,3 ± 2,7
Limo (%)	25	26,9 ± 2,8	33,8 ± 1,6	30,8 ± 1,8	26,4 ± 2,9	26,6 ± 4,5
Arcilla (%)	25	39,5 ± 2,2	32,8 ± 2,9	35,4 ± 2,1	33,8 ± 2,1	37,1 ± 2,9
Textura	25	F-a	F-a	F-a	F-a	F-a
Carbonatos (%)	25	8,45 ± 1,32	7,02 ± 0,54	9,87 ± 2,26	8,68 ± 1,49	8,97 ± 1,23
CE -p/v: ½- (dS m^{-1})	25	0,24 ± 0,13	0,26 ± 0,14	0,20 ± 0,10	0,19 ± 0,03	0,23 ± 0,03
pH -H_2O; p/v: ½-	25	8,03 ± 0,12	7,96 ± 0,14	7,95 ± 0,13	7,95 ± 0,13	7,91 ± 0,10
CIC ($cmol_c$ kg^{-1})	25	29,5 ± 0,9	23,3 ± 3,3	28,3 ± 1,0	26,0 ± 0,5	22,1 ± 3,5
PSI (%)	25	0,68 ± 0,46	0,27 ± 0,11	0,13 ± 0,01	0,12 ± 0,01	0,18 ± 0,03
CAU (%)	25	10,1 ± 1,8	11,9 ± 0,2	10,7 ± 0,5	10,7 ± 0,9	10,6 ± 1,5

Abreviaturas: N: número de muestras; F-a: textura franco-arcillosa; CE: conductividad eléctrica; p/v: relación peso (suelo) / volumen (agua); CIC: capacidad de intercambio catiónico; PSI: porcentaje de sodio intercambiable; CAU: cantidad de agua útil para las plantas (diferencia entre el contenido de agua a capacidad de campo: Ψ=-0,03 MPa y al punto de marchitamiento permanente: Ψ=-1,50 MPa).
Determinaciones realizadas siguiendo los métodos estandarizados descritos en MAPA (1994).

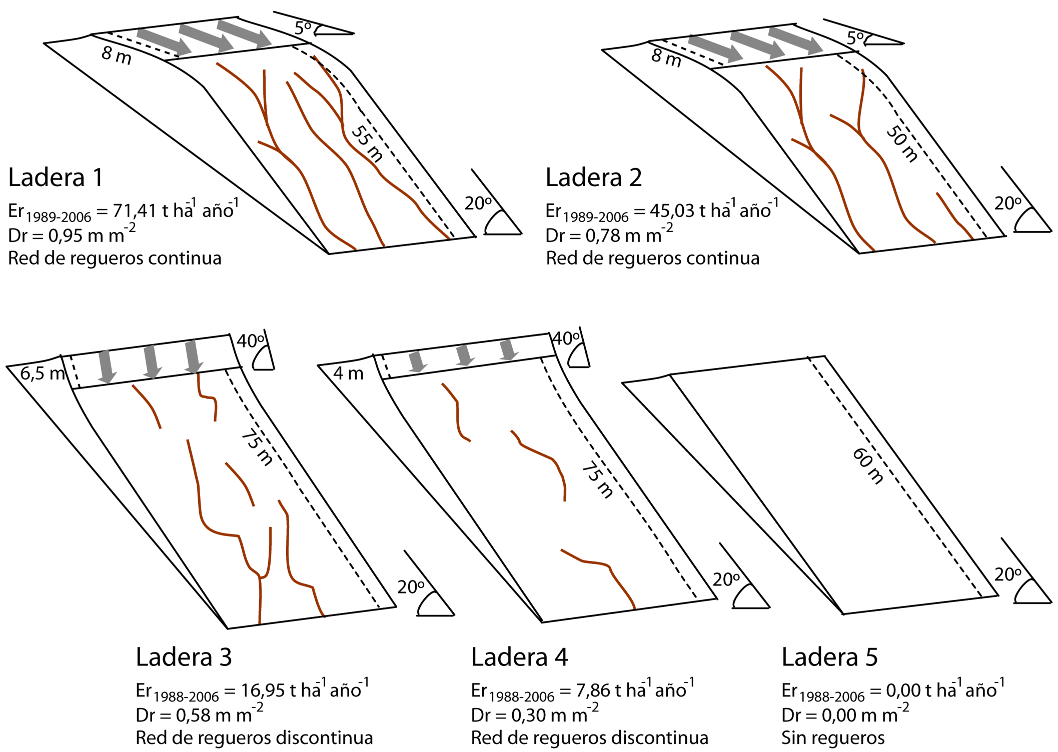

Figura 2.2. Esquema geomorfológico de las cinco laderas experimentales con representación gráfica de las áreas de contribución de escorrentía presentes en la cabecera de las mismas y el desarrollo asociado de las redes de regueros. Anotaciones: Er: tasa de erosión en regueros (determinada a partir de las dimensiones de las redes de regueros, siguiendo la metodología propuesta por Morgan, 1997); Dr: densidad de regueros (longitud lineal de regueros -m- por unidad de superficie -m^2-).

mite la generación de un patrón de rugosidad que aumenta la capacidad superficial de almacenamiento de agua y reduce la velocidad de los flujos de escorrentía. No se aplicaron enmiendas de tipo químico u orgánico. Las operaciones de revegetación consistieron en la siembra de una mezcla de semillas de gramíneas y leguminosas perennes (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis*, *Lolium perenne*, *Medicago sativa* y *Onobrychis viciifolia*).

Las cinco laderas difieren fundamentalmente en el diseño geomorfológico de sus cabeceras, presentando áreas de contribución de escorrentía de diferente tamaño. Así, un área desprovista de vegetación, de baja pendiente (4-6º) y considerable longitud (6-9 m) aparece en la parte alta de dos de las laderas experimentales (laderas 1 y 2). Del mismo modo, un área desnuda, de elevada pendiente (40º) y menor longitud (3-7 m) aparece en la parte alta de otras dos laderas experimentales (laderas 3 y 4). La última de las laderas experimentales (ladera 5) no presenta áreas de contribución ligadas a su cabecera. La presencia de estas áreas en la cabecera de las laderas ha favorecido el desarrollo temprano de procesos de erosión con formación de regueros de diferente intensidad (Figura 2.2), condicionando su evolución a largo plazo. La degradación a lo largo del tiempo de las condiciones hidrológicas de los suelos en estas laderas reguerizadas fue caracterizada por Nicolau (1996; 2002), mediante experimentación con lluvia simulada. Así, en la ladera 2, entre el tercer y séptimo año tras su restauración, los coeficientes de escorrentía aumentaron un 43%, mientras que las tasas finales de infiltración se redujeron un 21%. Durante este periodo y a escala de ladera, las tasas de erosión se mantuvieron en niveles muy altos (50-160 t ha^{-1} $año^{-1}$), ocasionando el desmantelamiento de la rugosidad superficial generada por las operaciones de labrado transversal, la formación de una red de regueros densa y el fracaso en el establecimiento y desarrollo de la vegetación introducida (5-15% cubierta vegetal).

Tras casi veinte años de evolución asociada a procesos de erosión en regueros de diferente intensidad (desde 0 t ha^{-1} $año^{-1}$ en el caso de ladera 5 a 71 t ha^{-1} $año^{-1}$ en el caso de la ladera 1) se pueden encontrar diferencias importantes en el desarrollo de la vegetación y de algunas características asociadas del suelo (materia orgánica, nitrógeno total, densidad) y su superficie (rugosidad), siguiendo un gradiente de degradación (Tabla 2.2; Imagen 2.3).

Aunque estas cinco laderas no constituyen réplicas perfectas y pueden existir algunos factores que se escapen al conocimiento obtenido por las investigaciones realizadas, constituyen un escenario ambiental excepcional para el análisis de los efectos ecológicos de la erosión hídrica superficial en sistemas restaurados de ladera.

Tabla 2.2. Principales diferencias entre laderas en atributos de la vegetación y del suelo (media ± desviación).

	N	Ladera 1	Ladera 2	Ladera 3	Ladera 4	Ladera 5
Suelo						
Materia orgánica [1] (%)	25	$0,58^{a} \pm 0,20$	$0,56^{a} \pm 0,23$	$1,27^{ab} \pm 0,35$	$1,46_{ab} \pm 0,83$	$2,00^{b} \pm 0,74$
Nitrógeno total [1] (%)	25	$0,04^{a} \pm 0,01$	$0,03^{a} \pm 0,01$	$0,07^{ab} \pm 0,02$	$0,07_{ab} \pm 0,04$	$0,10^{b} \pm 0,04$
D. del suelo [2] (g cm^{-3})	75	$1,51^{a} \pm 0,14$	$1,49^{a} \pm 0,12$	$1,39^{a} \pm 0,17$	$1,39^{a} \pm 0,12$	$1,23^{b} \pm 0,17$
Índice de rugosidad [3]	60	$1,04^{a} \pm 0,01$	$1,05^{a} \pm 0,01$	$1,10^{a} \pm 0,01$	$1,15^{a} \pm 0,03$	$1,23^{a} \pm 0,01$
Vegetación						
Cubierta [4] (%)	150	$1,1^{a} \pm 2,0$	$8,2^{a} \pm 5,5$	$27,8^{b} \pm 9,9$	$44,3^{bc} \pm 16,2$	$59,4^{c} \pm 20,8$
Biomasa [5] (g m^{-2})	30	$8,6^{a} \pm 12,2$	$29,5^{ab} \pm 20,5$	$65,3^{ab} \pm 36,9$	$158,8^{b} \pm 60,8$	$239,7^{b} \pm 95,3$
Necromasa [5] (g m^{-2})	30	$0,3^{a} \pm 0,5$	$9,8^{ab} \pm 4,7$	$10,7^{ab} \pm 17,9$	$23,8^{b} \pm 18,3$	$34,9^{b} \pm 17,8$

Abreviaturas: N: número de muestras; D.: densidad.

[1] Determinado usando métodos estandarizados descritos en MAPA (1994).

[2] Medido en cilindros inalterados de suelo (3 cm alto por 5 cm diámetro) recogidos de la superficie.

[3] Tortuosidad superficial (*sensu* Kamphorst *et al.*, 2000) en la dirección de la pendiente de la ladera; determinado a partir de levantamientos realizados con un microperfilador topográfico de 1 de longitud y 2 cm de resolución.

[4] Estimado visualmente en quadrats de 0.25 m^{2} en junio de 2006.

[5] Peso seco (60ºC, 72h) de muestras aéreas recogidas en quadrats de 0.25 m^{2} en junio de 2006.

Valores medios con letras diferentes (a-c) dentro de una misma fila difieren significativamente a α=0,05. Analizado usando Kruskal-Wallis ANOVA y *post-hoc* Mann-Withney.

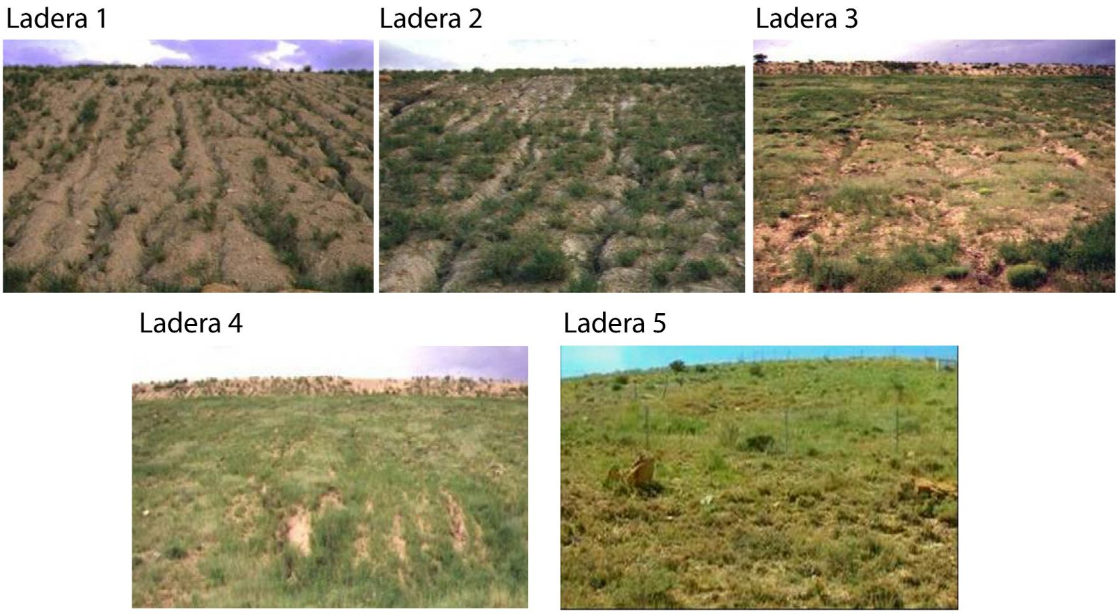

Imagen 2.3. Aspecto visual de las cinco laderas que componen el Área experimental Utrillas.

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Capítulo 3

Capítulo 3

Sucesión ecológica en laderas restauradas derivadas de la minería del carbón a cielo abierto en ambiente mediterráneo-seco

Este capítulo reproduce el texto del siguiente manuscrito:

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.

Resumen

Las restauraciones mineras desarrolladas durante los últimos 30 años en la cuenca lignitífera de Teruel han obtenido en general resultados poco exitosos. Este trabajo aborda el análisis de las trayectorias sucesionales y la identificación de las principales fuerzas directoras que controlan la dinámica vegetal en laderas restauradas mineras, con el objeto de mejorar los procedimientos de restauración. 87 laderas restauradas mineras de diferentes edades y tratamientos de restauración fueron clasificadas y posteriormente caracterizadas en base a diferentes variables relacionadas con la vegetación presente, topografía, técnicas de restauración, perturbaciones locales e intensidad de los procesos erosivos. Las diferentes trayectorias sucesionales, así como los factores, mecanismos y procesos implicados en éstas fueron inferidos a partir de análisis de gradientes. Se ha identificado una gran variedad de comunidades vegetales y trayectorias sucesionales. Las principales fuerzas directoras de estas trayectorias están representadas por las condiciones iniciales (calidad del suelo y tratamientos de revegetación) y el contexto ambiental (condiciones climáticas y distancia a las fuentes de propágulos) de las laderas restauradas. Asimismo, los procesos de erosión del suelo, desencadenados por flujos de escorrentía procedentes de estructuras superiores a las laderas, han sido identificados como un factor determinante para la evolución de la vegetación en estos ambientes secos. Otras perturbaciones de carácter local (pastoreo, enfermedades fúngicas) actúan sobre las comunidades vegetales favoreciendo la transición desde estados dominados por herbáceas no-nativas sembradas hacia comunidades más diversas que incorporan diferentes especies de matorral. Finalmente, se sugiere una serie de recomendaciones prácticas para el diseño y ejecución de futuros proyectos de restauración en ambientes mineros.

Palabras clave: ambientes secos, cuenca lignitífera de Teruel, erosión, inhibición, perturbaciones locales, restauración.

Vegetation succession in reclaimed coal-mining slopes in a Mediterranean-dry environment

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Abstract

Mining reclamation results obtained in the Teruel coalfield (Mediterranean-dry Spain) during the last 30 years have been quite limited. In order to improve restoration operations we conducted a study to analyse the trajectories of ecological succession and identify the main driving forces that control vegetation dynamic in reclaimed artificial slopes. A total of 87 slopes of different ages and restoration treatments were classified and characterized after recording different variables related to topography, restoration techniques, vegetation, local disturbances and soil erosion. Successional trends were inferred from gradient analysis as well as the factors, mechanisms and processes implied. We found a wide variety of plant communities and successional trajectories. Initial conditions (soil quality and revegetation treatments) as well as the environmental scenario (climatic conditions and vicinity of preserved propagule sources) were the main driving forces directing vegetation succession. Soil erosion triggered by external run-on coming from surrounding structures was also identified as a key factor determining the evolution of vegetation in these dry environments. Other local disturbances (grazing and fungal pests) can favour vegetation transition in communities dominated by highly competitive non-native sown species to more diverse shrub communities. Some practical considerations for future reclamation projects are suggested.

Key words: drylands, inhibition, local disturbances, restoration, soil erosion, Teruel coalfield.

Introduction

Failures in reclamation have been common in spite of the significant development of mining reclamation techniques during the last decades (Haigh, 2000); consequently, there are still poorly understood aspects (Plass, 2000). A deeper knowledge of the driving forces of community succession in reclaimed areas would improve their evolution towards ecosystems that contribute to provide territorial stability and facilitate the regeneration of fundamental ecological processes (Sänger and Jetschke, 2004).

In temperate areas, favourable conditions for spontaneous succession bring high potential for using it in reclamation processes (Prach and Pysek, 2001; Pietrzykowski and Krzaklewski, 2007). In these areas the main factors that control ecological succession are the following: regional meso-climatic differences, landscape factors related to the presence of preserved nearby vegetation, and local factors related to nutrient cycling and physico-chemical soil characteristics (Wiegleb and Felinks, 2001; Novak and Konvicka, 2006; Prach *et al*., 2007). Often, severe soil deficiencies and toxicity are the most relevant constraints (Bradshaw, 1997).

In reclaimed areas under dry climates, other factors related to water shortages and soil erosion may also affect vegetation dynamics (Whisenant, 2002; Martinez-Ruiz *et al*., 2007). Overland flow can be a driving force in these cases, since it redistributes soil particles (erosion and sedimentation) and water (runoff and soil moisture) at the slope scale (Lavee *et al*., 1998; Puigdefábregas *et al*., 1999). In man-made slopes, the incipient soil development favours overland flow run-off and limits rainfall infiltration (Ward *et al*., 1983; Guebert and Gardner, 2001). In some cases, extra overland flow runs (as run-on) into reclaimed slopes coming from the top and can promote soil erosion, with dramatic consequences for plant dynamics (Moreno-de las Heras *et al*., 2005).

Regardless of the particular environmental context, ecological succession occurring in restored environments is subjected to a wide range of contingencies, reducing the predictability of succession (Parker and Pickett, 1997). Some contingencies result particularly important in reclaimed mining ecosystems: mistakes in the execution of reclamation, changes in legislation, post-mining land uses, and the impact of local disturbances unconsidered during reclamation management (Nicolau and Moreno-de las Heras, 2005).

Surface mining in the Teruel coalfield (Mediterranean-dry Spain) started in 1976. During the last 30 years, 24 mines have been opened, affecting around 3000 hectares. During these three decades, reclamation practices have evolved as a consequence of legal requirements. Nevertheless, reclamation outcomes have been rather limited in most mines, and demand research in order to develop a successful restoration protocol (Mellado, 2006).

We studied the pattern and factors controlling vegetation succession in artificial slopes derived from opencast mining activities in the Teruel coalfield. Taking into account the presence of an environmental gradient related to climate continentality and landscape conservation, and the great heterogeneity of reclamation treatments in our study area, we expected a complex scenario expressed by different successional pathways in built slopes through time (about 20 years). We also expected to find that hydrologic dysfunction (mainly caused by erosion processes) and contingencies play a relevant role in vegetation dynamics, given the strong influence of macro-climate and the contingent nature of mining landscapes. The specific questions we wanted to answer are the following:

1. What is the relative importance of the environmental context and initial conditions on vegetation dynamics?

2. Can soil erosion be highlighted as a significant driving force for succession in these artificial slopes?

3. What is the role of contingences in the regional succesional pattern?

Materials and methods

Study area

This study was carried out in the Teruel coalfield (4900 km^2; Fig. 3.1), central-eastern Spain. The climate is Mediterranean. The regional moisture regime, classified as Mediterranean-dry (*sensu* Papadakis, 1966), is characterized by the concentration of rainy episodes in spring and autumn, and a period of summer drought.

A general environmental gradient related to meso-climatic differences and landscape conservation was defined according to geographic location within the study area:

1. In the eastern third, altitude is about 600-700 m.a.s.l. Climate continentality in this area is attenuated by maritime influence: mean annual temperature is 12.9-13.3 ºC, and annual thermal amplitude (difference between the annual absolute max. and min. temperature) is 43.2-45.8ºC. Mean annual precipitation and potential evapotranspiration are 413-478 and 727-740 mm respectively. Vegetation surrounding the areas affected by opencast coal mining is better preserved. It is formed by a mosaic of preserved forest patches (mainly dominated by *Pinus halepensis* and *Quercus ilex*) and abandoned terraces vegetated by shrubs (*Rosmarinus officinalis*, *Genista scorpius*, *Pistacea lentiscus*, *Thymus vulgaris* and *Dorycnium pentaphyllum*).

2. Altitude in the rest of the study area is about 700-1000 m.a.s.l. Climate is more continental: mean annual temperature is 11.0-14.0 ºC, and annual thermal amplitude is between 44.7-50.4ºC. Mean annual precipitation and potential evapotranspiration are 466-480 and 758-769 mm respectively. Vegetation surrounding the mines is more degraded, mainly because of overgrazing. Patches of natural forests are very scarce. Most of the landscape is covered by sparse shrub communities (dominated by *G. scorpius*) on abandoned terraces and cereal crops.

Regional natural soils (unaffected by mining) range from *Typic* or *Lithic Xerorthent* to *Calcic Xerochrept* (*sensu* Soil Survey Staff, 1998), with a low content of organic matter (<3%) and basic pH (Arranz, 2004).

Reclamation operations

Reclamation practices in the Teruel coalfield have progressed considerably in the last decades. Landform design has evolved from the oldest restored landscapes based on

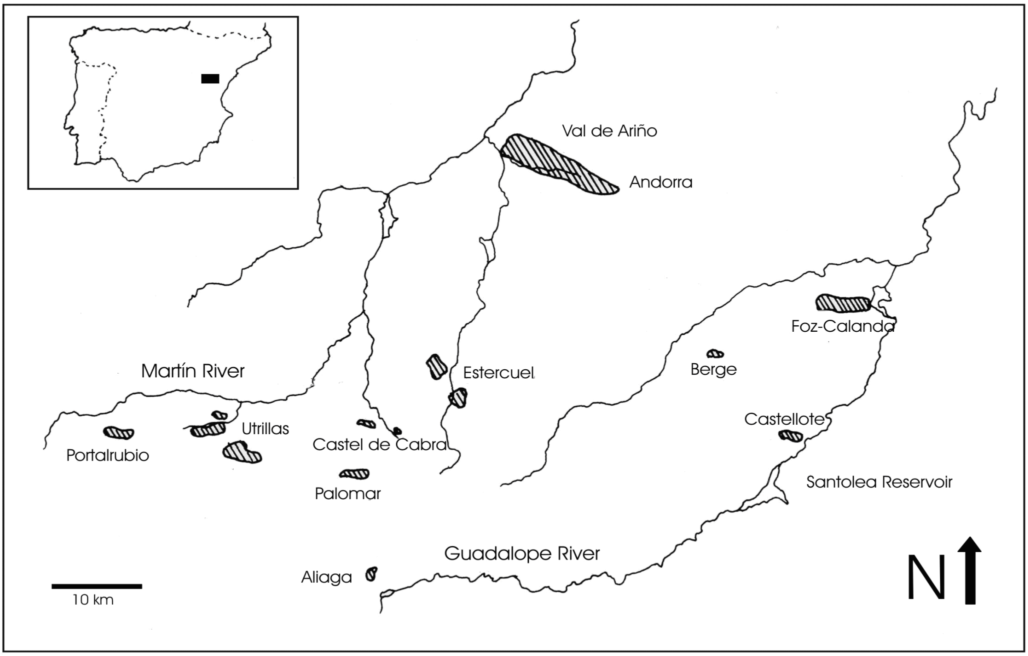

Figure 3.1. Geographic location of opencast coal mining in the Teruel coalfield. Stripes indicate areas affected by opencast coal mining.

platforms, banks and ditches to catchments structured by gentle slopes and watercourses (Nicolau, 2003). Soil management practices tend to favour the use of topsoil instead of overburden materials to cover the new forms. Nevertheless, in cases where revegetation is undertaken, a general mixture of non-native herbaceous grasses and legumes is sown, and only in a few cases this is followed by a further plantation of shrubs and/or tree species (see Appendix 3.A for species introduced by revegetation operations).

Field sampling

87 artificial slopes of different ages and reclamation treatments were sampled in two different field campaigns separated by a 15-year period (Table 3.1). During spring 1987 and 1988, information about 51 recently built slopes (0-6 years) was recorded. 36 older artificial slopes (10-20 years) were sampled in spring 2002 and 2003. Six slopes sampled in the second campaign coincide with slopes sampled in the first campaign (Table 3.1).

Common variables related to topography, such as steepness (º) and slope length (m) were measured. Likewise, the distance from natural seed sources -unaltered by mining- was recorded and transformed into a dummy variable (<200 and >800 m, as no slope between 200 and 800 m was sampled due to accessibility problems).

Each slope was divided into three 30 m cross-slope equidistant transects. Nine equidistant 0.25 m^2 plots were laid in each transect. For each plot, canopy cover (%) of

Table 3.1. Location of sampled slopes.

Localization	Sampled slopes		
	1987-88 field campaign	2002-03 field campaign	Coincident sampled slopes[1]
Utrillas	8	22	
Estercuel	12	6	2
Castellote	10	8	4
Palomar	10		
Val de Ariño	11		

[1] Slopes sampled during both field campaigns.

each species was recorded. This sampling procedure for vegetation survey has been successfully tested in reclaimed mining slopes of Mediterranean-dry Spain, encompassing more than 90% of species (Martinez-Ruiz *et al.*, 2007). Mean species composition per slope was calculated from data recorded at the plot scale and used for statistical analyses. The proportion of total vegetation cover (%), as well as two diversity indices (Shannon's index-H and Species richness-S), were calculated from mean slope species composition. In the same way, the relative abundance of species introduced by sowing (%) was calculated as the ratio between canopy cover of all sown species and total vegetation cover.

Three composite soil samples (each sample formed by six homogeneously mixed sub-samples, randomly distributed in each parallel transect) were taken from the first 15 cm of the soil profile in each slope. Soil parameters were analyzed following the standardized methods proposed by the Spanish Ministry of Agriculture (MAPA, 1994). Thus, stoniness (%) was determined as the content of soil particles >2 mm; textural particle size distributions were analysed following the Bouyoucos method and USDA classification; soil organic matter (%) and total nitrogen (%) were determined using the Walkley-Black and the Kjeldahl's methods respectively; and soil pH was determined in a 1/2 (weight/volume) aqueous solution with a Crison® mod.2001 pH-meter. Similarly to vegetation data, mean soil properties for each slope were calculated from the three composite soil samples and used for statistical analyses.

Every rill section with depth >1 cm was counted and measured (width and depth) for each transect. Mean number of rills per transect in each slope was determined. Rill erosion rate was calculated from the rill network dimensions following the methodology described by Morgan (1995). Variations of rock fragment cover can reflect variations in past erosion rates, due to the accumulated loss of fine soil particles from the soil surface, as a consequence of sheet erosion (Poesen *et al.*, 1998). In the same way, variations in rock fragment cover can also reflect spatial variations of rock fragment contents in the bulk soil. An accumulated sheet erosion index (ASEI), proportional to accumulated sheet erosion (soil particle loss due to sheet erosion since slope construction) and independent from the rock fragment content in soil was used. It was calculated as the quotient between mean rock fragment cover of the soil surface (calculated from rock fragment cover measured in 0.25 m^2 plots) and mean soil stoniness.

Information about restoration treatments (slope age, type of substrate, revegetation operations, as well as the presence of overland-flow contributing structures at the top of slopes and functional up-slope protective structures from run-on) was recorded as dummy variables by means of direct observation and the inspection of mining operation registers.

The occurrence of contingent local disturbances (grazing and fungal diseases) was also recorded as a dummy variable related to presence of sheep grazing signs (grazed vegetation, sheep wool and dung pellets), and fungal pests (fungal disease stains on leafs and micelle presence in stems and roots) on plot vegetation. Specifically, we found one species (*Medicago sativa*) highly affected by fungal attacks in some particular slopes.

Data analysis

To define the plant communities, slopes were grouped based on classification analysis and the identification of characteristic species (Velazquez and Gómez-Sal, 2007). TWINSPAN analysis (Hill, 1979) was performed to classify plant communities using a matrix with the slope plant composition. Three bare slopes from a total of 87 were excluded for analyses. Species present in less than 5% of slopes were eliminated to reduce erratic results (Sokal and Rohlf, 1995). Setup parameters of the classification analysis were: 3 as maximum number of indicators per division, 4 as maximum level of division and 10 as minimum group size for division. Indicator Species Analysis (ISA, Dufrene and Legendre, 1997) was used to establish the characteristic species of each group.

In order to characterize plant communities and check whether significant differences existed, non-parametric Kruskall-Wallis tests and *post hoc* Mann-Whitney U tests were performed. To control the rate of spurious significance of related tests sequential Bonferroni corrections (Rice, 1989) were used (α= 0.01 for main tests and α= 0.05 for *post-hoc* tests).

The classified and characterised plant communities were reflected by a Detrented Correspondence Analysis (DCA, Hill and Gauch, 1980), performed with the same matrix, to define the underlying ecological structure (van Groenewoud, 1992). Relevant passive variables (environmental factors as well as plant and erosion traits) were projected in DCA configuration as vectors to interpret the gradients expressed by DCA axes. To avoid redundancy in gradient interpretation, passive variables fitted to DCA configuration were previously selected within significant variables for community characterization by cross-correlation (Vogiatzakis *et al*., 2003).

PC-ORD Version 4 (McCune and Mefford, 1999) was used for TWINSPAN and ISA. The Vegan package from the R system (Oksanen *et al*., 2007) was used for DCA and vector fitting. STATISTICA 6.0 (Statsoft, 2001) was used for the complementary statistical analyses.

Results

Plant community classification and characterization

Considering floristic composition (83 species appeared in more than 5% plots of the 84 vegetated slopes; Appendix 3.A), and after the grouping of slopes according to the TWINSPAN analysis and their characteristic species (established using Indicator Species Analysis), eight different plant communities were identified (Table 3.2). These plant communities were related to both the general restoration treatments applied and environ-

Table 3.2. General description of the plant community groups derived from TWINSPAN analysis.

Code	N	Restoration treatments	Environmental factors	Plant traits	Erosion traits	Characteristic Species	p
CT1	18		↓pH	↓↓ Vc ↓↓ S ↓↓ H	↑↑Re	*Polygonum aviculare* *Iberis sp*	0.007 0.019
CT2	11	Revegetated	Continental Inf ↓Age	↓Vc ↓S	↑Re	*Lolium perenne* *Onobrychis viciifolia*	0.025 0.042
CT3	10	Topsoiled	Maritime Inf ↓Age	↓Vc ↓S	↑Re	*Medicago minima* *Silene muscipula* *Raphanus raphanistrum* *Erodium cicutarium*	0.004 0.004 0.005 0.012
CT4	4	Topsoiled	Maritime Inf ↑Age	↑Vc ↑S ↑H		*Coronilla scorpioides* *Helicrysum stoechas* *Astragalus hamosus* *Artemisia campestris*	0.001 0.001 0.002 0.002
CT5	17	Revegetated	Continental Inf Run-on.	↓↓Vc ↓↓S ↓↓H ↑Sspp	↑↑Re ↑Se	*Medicago sativa*	0.010
CT6	14	Revegetated	Continental Inf ↑Age	↑Vc ↓S ↑Sspp		*Festuca arundinacea* *Sonchus asper* *Filago vulgaris* *Anacyclus clavatus*	0.001 0.006 0.010 0.012
CT7	6	Revegetated	Continental Inf Local disturbances ↑Age	↑Vc ↑S ↑H		*Thymus vulgaris* *Xeranthemum annuum* *Santolina chamaecyparissus* *Plantago lanceolata*	0.002 0.002 0.003 0.003
CT8	4	Topsoiled	Maritime Inf Closed to Nss ↑Age	↑Vc ↑S ↑H		*Rosmarinus officinalis* *Genista scorpius* *Dorycnium pentaphyllum* *Plantago albicans*	0.002 0.002 0.005 0.008

General abbreviations: N: number of representative slopes in each community group; *p*: Monte Carlo test significance for characteristic species assigned by the Indicator Species Analysis.
Abbreviations for environmental factors: Inf: influence; Nss: natural seed sources (unaltered by mining).
Abbreviations for plant traits: Vc: total vegetation cover; S: species richness; H: Shannon's index; Sspp: relative sown species abundance.
Abbreviations for erosion traits: Re: rill erosion; Se: accumulated sheet erosion.
↑= high; ↑↑very high; ↓ low; ↓↓= very low (based on the quantitative differences summarized in Appendix 3.B).

mental factors (meso-climate and landscape context influence), and were very different in terms of plant and erosion traits (Table 3.2; Appendix 3.B).

A very sparse (almost bare) and poor community (CT1; Table 3.2) was identified on unreclaimed slopes, characterised by the presence of acid overburden substrates.

On the eastern mining sites (specifically the Castellote mining site), where climate is less continental and surrounding vegetation is better preserved, three different plant communities (CT3, CT4 and CT8; Table 3.2) were identified on unrevegetated slopes covered with topsoil from abandoned agricultural terraces. These three communities differed basically in vegetation cover, diversity and age; the oldest ones (CT4, CT8) were better covered and more diverse, and in the case of the CT8 community (slopes <200 m from natural seed sources), its composition was characterised by the presence of different shrub species (*Genista scorpius*, *Rosmarinus officinalis* and *Dorycnium pentaphyllum*).

Four different plant communities (CT2, CT5, CT6 and CT7; Table 3.2) were identified on slopes sown with herbaceous seed blends on the western and central mining sites, where climate is more continental and surrounding areas are degraded by overgrazing. With respect to substrate, these groups were fairly heterogeneous as both overburden and topsoil were used to cover the slopes where these communities appeared. These four groups differed on vegetation complexity and degradation through erosion. Soil erosion processes were mainly triggered by the presence of active sources of overland flow at the top of these slopes ($F_{1, 84}$= 40.41; p<0.000). In fact, a highly eroded community type (CT5) was found on slopes of different ages where up-slope protective structures from run-on (channels and berms which should be designed to isolate slopes from other structures) were broken or absent. This community was characterised by a very low cover, vegetated by almost only one species, *Medicago sativa*, a sown perennial legume. Two less eroded herbaceous communities (CT2, CT6) were found, differing in age and vegetation cover. Soil surface was less covered by vegetation in the youngest group (CT2); the oldest one (CT6), better covered, was characterized by the massive presence of sown species (up to 70% of total cover), especially the legume *M. sativa*. Finally, a more diverse shrub community was identified (CT7) on old slopes with sings of sheep grazing on vegetation and/or fungal affection on *M. sativa* populations.

Gradient analysis: the underlying ecological processes

The identified plant communities were represented using DCA (Fig. 3.2), performed with the same data matrix as the classification analysis. The first correspondence axis (27.2% of explained variation) was positively linked with rill erosion and inversely with age and soil pH (Fig. 3.2). Likewise, the second axis (13.1% of explained variation) was positively linked with diversity and occurrence of local disturbances, and inversely linked with the distance from seed sources and relative sown species abundance (Fig. 3.2). Other factors (climate and conservation of surrounding landscape) may play a role in the gradient expressed by axis 2. Thus, CT3, CT4 and CT8 community type groups, which are formed by topsoiled and unsown slopes from the Castellote mining site in the eastern third of the studied area (lowest climatic continentality and surrounding vegetation better preserved), were distributed in DCA biplot along the second half of axis 2 (Fig. 3.2).

Discussion

Successional pattern and driving forces

A wide variety of plant communities and trajectories were identified in the Teruel coalfield (Fig. 3.3). This is expected in post-mining ecosystems, where vegetation dynamics can be considered a type of primary succession with low probability of vegetation convergence (Wiegleb and Felinks, 2001; Novak and Prach, 2003).

Based on gradient analysis and differences between structural and environmental features,

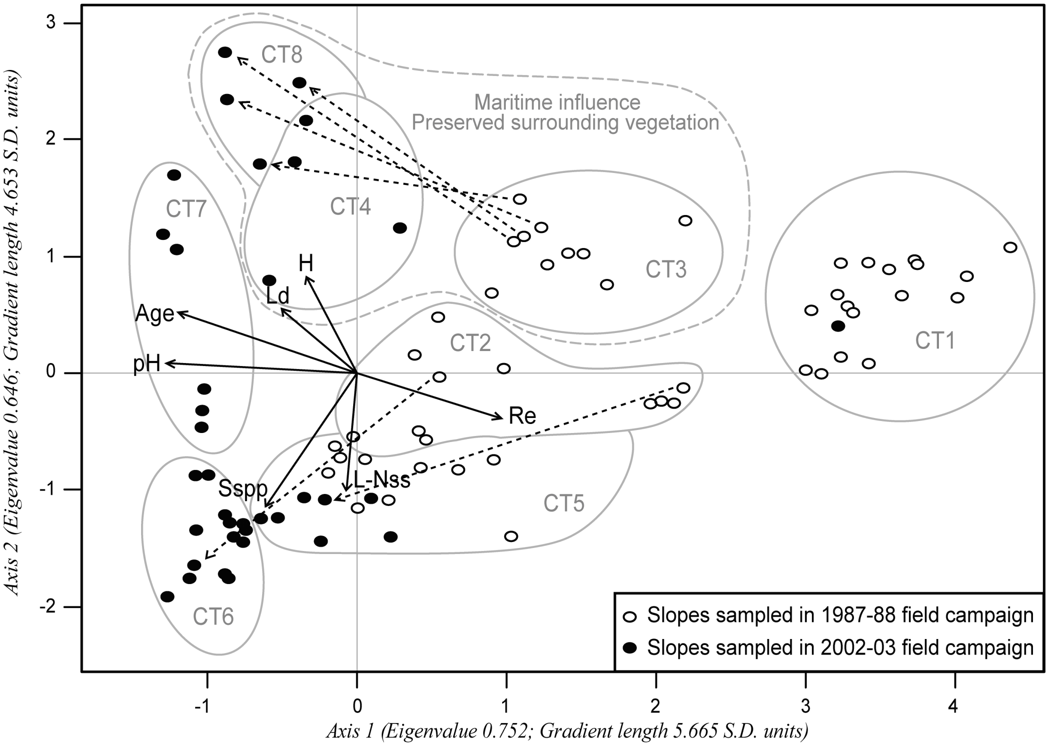

Figure 3.2. DCA biplot showing ordination of sampled slopes. Black solid vectors represent the strength and direction of significant correlations (α=0.01) between representative passive variables (environmental factors as well as plant and erosion traits) and ordination axes. Black dotted vectors represent the temporal transition of the six coincident sampled slopes. Solid gray borderlines group the community types derived from TWINSPAN analysis. The broken gray borderline groups slopes located in the Castellote mining site (eastern extreme of the environmental gradient), where climate is less continental and surrounding vegetation is better preserved. Abbreviations for passive variables: Re: rill erosion; H: Shannon's diversity index; Ld: local disturbances; Sspp: relative sown species abundance; L-Nss: length from seed sources.

three main pathways can be inferred (Fig. 3.3):

(a) On acid soils, plant establishment and community development are highly limited, directing the system towards very simple and poor states (CT1).

(b) Where the environmental conditions are less restrictive (lower continentality, better soil quality and vicinity of preserved natural forests) the transition from the initial community (CT3) to more diverse ones (CT4 and CT8) is always observed. In this case, the main driving force is the proximity of propagule sources.

(c) Where the environmental conditions are more restrictive, other driving forces become more relevant. These forces are related to initial conditions (communities based on competitive non-native species) and the occurrence of some disturbances as rill erosion, mainly triggered by external overland flow, or grazing and fungal pests. In very rilled slopes only a sparse and simple community (CT5) can develop. Where extra overland flow does not run into slopes, initial sown communities (CT2) progress into a persistent herbaceous community (CT6) dominated by a non-native sown legume (*Medicago sativa*). In this case, grazing by sheep and/or fungal pests drives the system towards a more diverse community, dominated by shrub species (CT7).

Environmental context and initial conditions

In the Teruel coalfield area, the divergent successional pattern is primarily ruled by both the initial conditions of mining slopes and the environmental gradient associated to climate and conservation of surrounding vegetation. Similar results have been found in central European man-made habitats, where the initial conditions of altered areas (specifically soil pH) and meso-climatic differences drive successional patterns (Prach *et al*., 2007).

Regarding the chemical soil characteristics, the temporal successional progress was related to soil pH levels (Fig. 3.2). In the study area, plant immigration, vegetation establishment and species replacement on acid sites were highly constrained (Fig. 3.3). This is probably due to the absence of appropriate species in nearby propagule sources (which grow on basic soils) and occasional toxicity associated to extremely acid substrates. Analogous restrictions to natural recovery associated to soil acidity and toxicity have been frequently reported in mining sites (Bradshaw, 1997; Prach and Pysek, 2001). In the Teruel coalfield, this problem mainly affects derelict mines which operated before the regulation

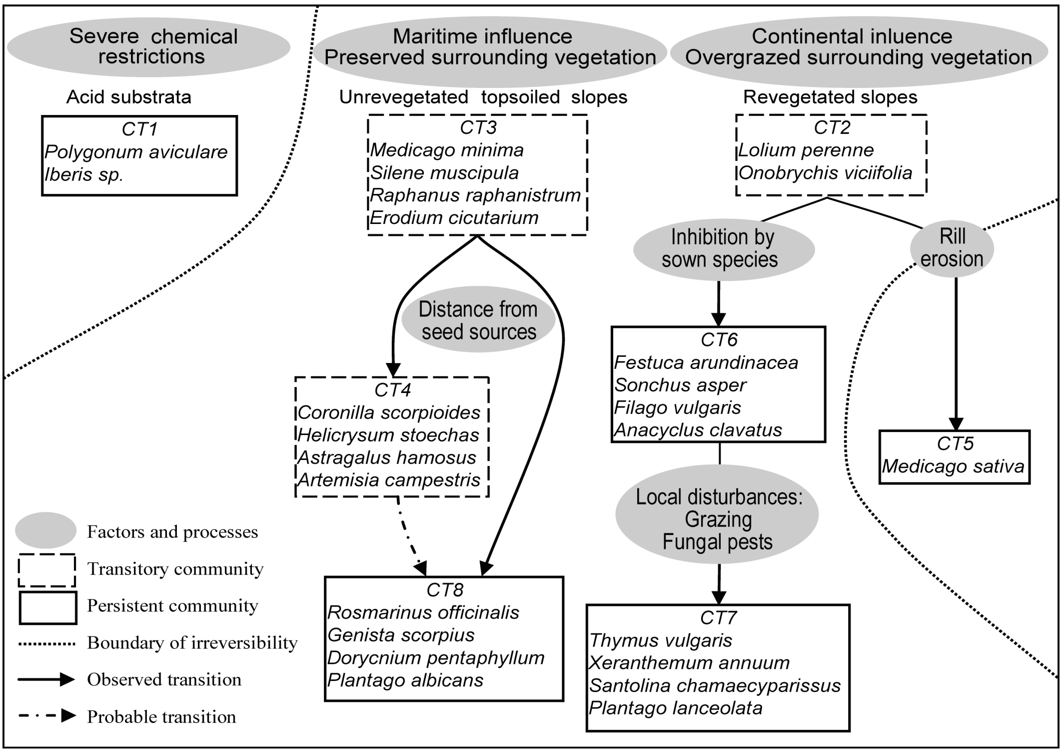

Figure 3.3. General scheme of succession in reclaimed artificial slopes in the Teruel coalfield. Displayed plant communities and characteristic species has been identified using TWINSPAN and Species Indicator Analysis. Community transitions as well as the relevant factors and processes involved has been inferred from community characterization and gradient analysis.

introduced by the first Spanish reclamation law in 1982 (Spanish Royal Decree, RD 2994/1982). Indeed, prior to the application of the cited regulation, no procedure of substrate selection to cover the altered surfaces was applied by local mining companies.

In the absence of serious restrictions caused by soil characteristics, climate and the presence of well-preserved surrounding vegetation were responsible for the different successional trends. In this way, all slopes located in the Castellote mining site (situated in the least continental area of the Teruel coalfield) were grouped towards the top of DCA axis 2. Such influences caused by meso-climatic differences have been generally attributed to variations in the species pool (Otto *et al*., 2006). Nevertheless, other factors could also play a role in successional differentiation and subsequent community transitions identified in this mining site. In fact, these slopes were reclaimed with good quality topsoil and were surrounded by preserved forest patches of *Pinus halepensis* and diverse shrub communities (*Rosmarino-Ericion multiflorae*; *sensu* Rivas-Martinez, 1987). This aspect is especially important for reclaimed mining sites, where community structure is usually limited by long distances from seed sources and reduced dispersal ability of species (Parrotta and Knowles, 2001; Wiegleb and Felinks, 2001; Novak and Konvicka, 2006). Indeed, the establishment of some shrub species characterised by limited barochorous seed dispersal (as *Genista scorpius* and *Dorycnium pentaphyllum*) was only possible in slopes nearest to preserved patches, resulting in a more diverse community (CT8, Fig. 3.3).

In more continental areas, a strong effect caused by the use of mixtures of non-native fast growing herbaceous species appeared. This is supported by DCA axis 2, which confronts the relative abundance of sown species with species diversity. These herbaceous mixtures were generally used to simply "paint slopes green" and to comply with legal requirements for soil erosion and water quality control. In consequence, succession was arrested in most of these revegetated slopes, leading to a persistent herbaceous community (CT6) mainly dominated by *Medicago sativa*. Similarly, it has been generally mentioned that revegetation with fast growing herbaceous species can prevent long-term vegetation development due to competition with spontaneous colonizers (Holl, 2002). This result emphasizes the need to modify the composition of revegetation seed mixtures using native species; these mixtures should include some successful colonizing shrub species which may be restricted by the availability and distance of the natural propagule sources.

The role of soil erosion

Soil erosion must be considered as a significant process for restored vegetation dynamics in the studied area. This is shown by the opposite relationship with successional advance (Fig. 3.2). Accordingly, soil erosion, and particularly rill erosion, has been highlighted as a key limiting factor for vegetation establishment and succession in both natural and reclaimed slopes (Nicolau and Asensio, 2000; Wang *et al*., 2007). In the studied slopes, rill erosion is frequently triggered because of the presence of overland flow contributing areas at the top of reclaimed slopes. Failures in the design, as the absence or collapse of up-slope diverting structures (channels and berms), allow overland flow run-on the slopes promoting soil erosion.

In the present study, severe rill erosion processes are associated with the development of a very sparse and simple community essentially composed of a few plants of *Medicago sativa* (Fig. 3.3). Similar communities have been documented in other severely eroded Mediterranean-dry environments, as a consequence of the increase of mechanical disturbance and water stress supported by vegetation (Guardia, 1995; Guerrero-Campo and Montserrat-Martí, 2004). Accordingly, other studies carried out in reclaimed slopes of the Teruel coalfield have pointed at the negative influence of rill erosion on plant colonization and establishment as a consequence of a drastic reduction in water availability (Moreno-de las Heras *et al*., 2005). The main mechanisms involved are the reduction of water infiltration by surface crust formation and soil surface roughness reduction, and the efficient evacuation of overland flow from the slopes by rill networks (Nicolau, 2002; Moreno-de las Heras *et al.*, 2007). The resprouting ability and deep rooting system of *Medicago sativa* could explain plant survival in these harsh conditions, lightening the impact of mechanical disturbance and water stress caused by soil erosion (Bell *et al*., 2007).

Implications of local disturbances

As expected, some contingent factors were involved in community changes. A positive connection between diversity increases and the occurrence of local disturbances (sheep grazing and fungal diseases) on dominant herbaceous populations is reflected in DCA axis 2 (Fig. 3.2). In fact, local disturbances has been stressed as important sources of successional change, controlling inhibition mechanisms (Pickett *et al*., 1987). In the present study, the transition from persistent herbaceous communities (CT6) to a more diverse shrub community (CT7) was related to the occurrence of both occasional fungal diseases in the population of *Medicago sativa*, and sheep grazing. The primary effect of both factors is the creation of gaps free of competition which enhance the chances of seed germination and plant establishment for new species, especially anemochorous shrubs (as *Santolina chamaecyparissus* and *Thymus vulgaris*). In addition to site control, grazing can also act as a vector of seed dispersal, since large quantities of seeds can be transported through seed retention in fur and the alimentary tract (Mitlacher *at al*., 2002). These results highlight the potential function of sheep grazing as a restoration tool in the studied area, bearing in mind the involved benefits from seed dispersal and the control of dominant species introduced by sowing.

Conclusions

The evolution of vegetation communities in reclaimed slopes in the Mediterranean-dry area of the Teruel coalfield showed a complex pattern in which three main trends can be identified: (i) very poor communities or even bare slopes on acid soils, (ii) spontaneous communities in which diversity increases with time in less continental areas surrounded by preserved vegetation, and (iii) revegetated communities in more continental areas surrounded by overgrazed vegetation, in which case, transitions are highly dependent on the occurrence of disturbances (rill erosion, grazing or fungal pests). Therefore, initial conditions (soil characteristics and revegetation treatments) and the environmental scenario (climate continentality and presence of surrounding preserved vegetation) appear as the main driving forces directing vegetation succession in reclaimed slopes.

Distance from seed sources is the key factor that directs community development in areas where environmental conditions are less restrictive. Erroneous revegetation practices and design mistakes (presence of overland flow contributing areas at the top of reclaimed slopes) have an important weight in the explanation of vegetation dynamics under restrictive environmental conditions. The use of non-native fast growing herbs can seriously constrain vegetation dynamics. In these cases, local disturbances (such as the occurrence of fungal diseases and, especially, sheep grazing on dominant herbaceous populations) can promote transitions to more diverse shrub communities. Soil erosion triggered by run-on coming from the top of reclaimed slopes constitutes a significant driving force for vegetation succession in reclaimed slopes in Mediterranean-dry environments. On intensively rilled slopes plant establishment was severely restricted, probably because of an increase in water stress and physical disturbance caused by accelerated soil erosion.

This work is not the result of an experimental design aimed at the study of all the possible combinations of factors involved on vegetation dynamics (i.e.: environmental scenario and initial conditions). Thus, this research would greatly benefit from experimental confirmation in the future. Nevertheless, we can draw the following practical considerations from our results: special attention must be placed in the selection of a substrate free of important physico-chemical restrictive factors (extreme pH and toxicity), up-slope protective structures (channels and berms) must be preserved to control run-on fluxes from the top of reclaimed slopes, and revegetation with fast growing allochtonous species must be avoided in order to prevent the inhibition of spontaneous colonization.

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Appendix 3.A . List of species identified.

Boraginaceae
- *Echium vulgare*
- *Myosotis arvensis*
- *Neatostema apulum*

Caryophyllaceae
- *Silene muscipula*
- *Silene nocturna*
- *Vaccaria hispanica*

Chenopodiaceae
- *Atriplex halimus**[1]
- *Atriplex sp*
- *Salsola kali*
- *Salsola vermiculata*

Cistaceae
- *Cistus clusii*
- *Helianthemum apenninum*
- *Helianthemum nummularium*

Compositae
- *Anacyclus clavatus*
- *Artemisia campestris*
- *Bellis perennis**
- *Carduus pycnocephalus*
- *Centaurea aspera**
- *Filago vulgaris*
- *Helichrysum stoechas*
- *Hieracium pilosella**
- *Mantisalca salmantica**
- *Pallenis spinosa*
- *Santolina chamaecyparissus*
- *Senecio vulgaris**
- *Sonchus asper*
- *Xeranthemum annuum*

Convolvulaceae
- *Convolvulus arvensis*

Crassulaceae
- *Sedum album**

Cruciferae
- *Alyssum alyssoides*
- *Diplotaxis erucoides*
- *Eruca vesicaria*
- *Iberis sp*
- *Matthiola fruticulosa**
- *Raphanus raphanistrum*
- *Sisymbrium orientale*

Euphoriaceae
- *Euphorbia serrata**

Geraniaceae
- *Erodium ciconium*
- *Erodium cicutarium*
- *Erodium malacoides**

Graminae
- *Aegylops geniculata*
- *Arrhenatherum elatius**
- *Avena sterilis*
- *Avenula bromoides**
- *Brachypodium retusum*
- *Bromus willdenowii**[1]
- *Bromus diandrus**
- *Bromus erectus*
- *Bromus hordeaceus*
- *Bromus rubens*
- *Bromus tectorum*
- *Cynodon dactylon*
- *Dactylis glomerata*
- *Desmazeria rigida*
- *Elymus hispidus*
- *Festuca arundinacea*[1]
- *Holcus lanatus*
- *Hordeum murinum*
- *Lolium perenne*[1]
- *Phleum pratense*
- *Sorghum bicolor**
- *Stipa iberica**
- *Stipa pennata**
- *Triticum aestivum**
- *Wangenheimia lima**

Labiatae
- *Lavandula angustifolia**
- *Marrubium supinum*
- *Rosmarinus officinalis*
- *Satureja sp**
- *Thymus vulgaris*

Leguminosae
- *Anthyllis cytisoides*
- *Argyrolobium zanonii*
- *Astragalus hamosus*
- *Coronilla scorpioides*
- *Dorycnium pentaphyllum*
- *Genista scorpius*
- *Hippocrepis glauca*
- *Hippocrepis comosa*
- *Medicago lupulina*
- *Medicago minima*
- *Medicago polymorpha*
- *Medicago sativa*[1]
- *Melilotus indica*
- *Melilotus officinalis*[1]
- *Onobrychis viciifolia*[1]
- *Ononis spinosa**
- *Psoralea bituminosa*
- *Scorpiurus muricatus*
- *Trifolium pratense**
- *Vicia villosa*

Linaceae
- *Linum narbonense**
- *Linum suffruticosum**

Malvaceae
- *Althaea hirsuta*
- *Malva hispanica**

Papaveraceae
- *Fumaria parviflora**
- *Papaver rhoeas*

Pinaceae
- *Pinus halepensis**[1]

Plantaginaceae
- *Plantago albicans*
- *Plantago lanceolata*
- *Plantago sempervirens*

Polygonaceae
- *Polygonum aviculare*

Primulaceae
- *Anagallis arvensis*
- *Androsace maxima*
- *Coris monspeliensis**

Ranunculaceae
- *Aquilegia vulgaris**

Rosaceae
- *Potentilla crantzii*
- *Rosa canina**
- *Sanguisorba minor*

Rubiaceae
- *Galium verum*

Sanatalaceae
- *Thesium humifusum*

Umbelliferae
- *Bupleurum baldense*
- *Daucus carota**
- *Eryngium campestre*

Nomenclature follows Tutin *et al.* (1964-1980).
* Species present in less than 5% of the slopes.
[1] Species introduced by revegetation operations.

Appendix 3.B. Comparison of plant, environmental and erosion traits associated to the community groups derived from TWINSPAN analysis (Mean±SD).

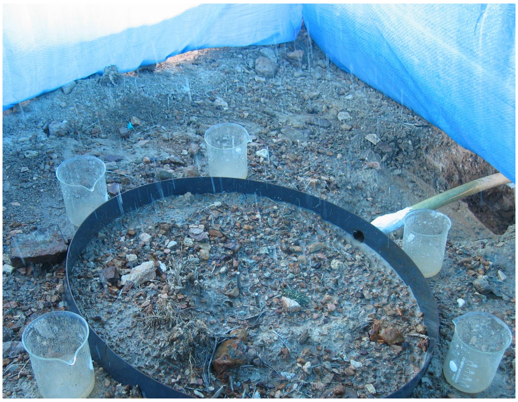
	Plant community							
	CT1	CT2	CT3	CT4	CT5	CT6	CT7	CT8
Plant traits								
Veget cover (%)	5.6 ± 5.1^{a}	21.3 ± 7.0^{b}	21.1 ± 12.3^{b}	34.6 ± 2.7^{bc}	13.5 ± 8.9^{ab}	38.2 ± 18.6^{bc}	51.2 ± 10.8^{c}	29.5 ± 9.3^{bc}
Litter cover (%)	3.2 ± 2.4^{a}	10.2 ± 1.4^{b}	10.3 ± 2.2^{b}	15.8 ± 2.7^{b}	4.7 ± 3.5^{a}	13.3 ± 7.6^{b}	11.2 ± 3.8^{b}	16.7 ± 3.1^{b}
Species richness - S	2.6 ± 2.0^{a}	8.2 ± 3.4^{b}	12.9 ± 6.1^{bc}	27.3 ± 4.5^{cd}	5.7 ± 5.0^{ab}	16.6 ± 3.8^{c}	30.7 ± 9.4^{d}	25.3 ± 7.3^{c}
Shannon's index - H	0.6 ± 0.6^{a}	1.5 ± 0.4^{b}	2.0 ± 0.5^{b}	1.67 ± 0.4^{b}	0.8 ± 0.7^{ab}	1.5 ± 0.5^{b}	2.3 ± 0.6^{b}	2.0 ± 0.3^{b}
Relative Sspp (%)	0.0 ± 0.0^{a}	47.2 ± 17.1^{c}	0.0 ± 0.0^{a}	0.0 ± 0.0^{a}	75.9 ± 28.9^{c}	71.4 ± 17.5^{c}	26.9 ± 19.3^{bc}	0.0 ± 0.0^{a}
Environmental variables								
Age	4.0 ± 1.1^{a}	2.7 ± 0.5^{a}	3.1 ± 1.4^{a}	19.0 ± 0.0^{c}	10.4 ± 6.9^{a}	12.4 ± 1.6^{b}	14.8 ± 0.4^{b}	19.0 ± 0.0^{c}
Steepness (º)	27.3 ± 8.1^{ab}	24.1 ± 4.9^{b}	33.8 ± 3.0^{a}	33.3 ± 0.6^{a}	24.7 ± 7.8^{ab}	17.5 ± 1.6^{b}	16.7 ± 0.8^{b}	32.8 ± 4.0^{a}
Length (m)	21.6 ± 4.5^{a}	32.1 ± 13.6^{ab}	22.8 ± 4.2^{a}	24.7 ± 8.1^{a}	40.9 ± 17.7^{ab}	43.9 ± 19.0^{ab}	55.9 ± 11.6^{b}	24.5 ± 7.8^{a}
Stoniness (%)	21.7 ± 5.2^{b}	32.5 ± 11.7^{ab}	39.9 ± 14.1^{ab}	41.0 ± 8.2^{a}	23.6 ± 10.4^{b}	21.7 ± 8.4^{b}	34.8 ± 9.1^{ab}	45.6 ± 7.4^{a}
Sand (%)	34.8 ± 9.8^{a}	39.4 ± 7.1^{a}	37.9 ± 6.2^{a}	44.0 ± 3.1^{a}	36.9 ± 11.6^{a}	41.7 ± 3.6^{a}	43.7 ± 4.5^{a}	39.1 ± 3.6^{a}
Silt (%)	32.1 ± 7.1^{a}	29.1 ± 3.8^{a}	30.9 ± 3.1^{a}	29.1 ± 2.9^{a}	26.7 ± 7.1^{a}	24.6 ± 3.3^{a}	24.0 ± 2.5^{a}	32.3 ± 1.7^{a}
Clay (%)	33.1 ± 9.2^{a}	31.5 ± 6.0^{a}	31.6 ± 8.2^{a}	26.8 ± 0.8^{a}	36.4 ± 8.4^{a}	33.5 ± 2.2^{a}	32.5 ± 2.8^{a}	28.6 ± 2.4^{a}
pH	5.5 ± 1.6^{a}	7.8 ± 0.6^{bc}	7.4 ± 0.9^{bc}	8.5 ± 0.1^{c}	7.8 ± 0.4^{b}	8.4 ± 0.1^{bc}	8.3 ± 0.1^{bc}	8.6 ± 0.1^{c}
Organic matter (%)	2.1 ± 1.7^{a}	1.3 ± 1.5^{a}	1.0 ± 0.4^{a}	1.4 ± 0.1^{a}	1.3 ± 0.7^{a}	1.6 ± 0.6^{a}	2.3 ± 0.8^{a}	1.0 ± 0.3^{a}
Total nitrogen (%)	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}	0.1 ± 0.1^{a}
Erosion traits								
ASEI	0.6 ± 0.2^{b}	0.6 ± 0.4^{b}	0.6 ± 0.2^{b}	1.1 ± 0.2^{a}	1.7 ± 1.3^{a}	0.8 ± 0.5^{a}	0.8 ± 0.3^{a}	1.3 ± 0.3^{a}
Rills in transects	35.8 ± 10.3^{b}	22.9 ± 15.0^{b}	12.9 ± 12.5^{b}	0.0 ± 0.0^{a}	37.0 ± 12.9^{b}	3.6 ± 5.2^{ab}	3.8 ± 9.4^{ab}	1.3 ± 1.9^{a}
Re rate ($t\ ha^{-1} year^{-1}$)	33.2 ± 18.1^{c}	11.1 ± 9.1^{b}	19.8 ± 16.7^{b}	0.0 ± 0.0^{a}	28.8 ± 15.3^{c}	3.1 ± 5.2^{ab}	4.3 ± 10.6^{ab}	2.1 ± 4.2^{a}

Abbreviations for variables: Veget: green vegetation; Sspp: sown species abundance; ASEI: accumulated sheet erosion index; Re: rill erosion.

Kruskal-Wallis and *post-hoc* U tests were applied to detect significant differences between community types.

Different letters (a-c) within rows indicate differences at α=0.05 after the application of sequential Bonferroni corrections.

Capítulo 4



Capítulo 4

Efecto de la cubierta vegetal sobre la hidrología de los suelos restaurados mineros bajo clima mediterráneo-continental

Este capítulo reproduce el texto del siguiente manuscrito:

Moreno-de las Heras, M., Merino-Martín, L., Nicolau, J.M. En prensa. Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate. *Catena*; DOI: 10.1016/j.catena.2008.12.05.

Resumen

La cubierta vegetal juega un papel fundamental en la estabilización y restauración de las áreas alteradas. En el caso de las restauraciones mineras, el estudio de las relaciones existentes entre la cubierta vegetal y la hidrología del suelo tiene una relevancia especial tanto para el diseño de las operaciones de restauración como para la evaluación de los resultados obtenidos. Este tipo de análisis tiene una importancia clave en ambientes mediterráneo-continentales, donde las condiciones climáticas limitan las posibilidades de desarrollo de cubiertas vegetales espacialmente continuas. En este trabajo se analiza el efecto de la cobertura de herbáceas en la respuesta hidrológica a la lluvia simulada (63 mm h^{-1}; 0.24 m^2) de suelos restaurados derivados de la minería del carbón a cielo abierto (un estéril no salino de textura franco-arcillosa). En total se han realizado 75 experimentos de lluvia simulada. Para controlar la influencia de las fluctuaciones estacionales climáticas en las respuestas hidrológicas, los experimentos se han llevado a cabo en tres momentos diferentes a lo largo del año (final de invierno, verano y otoño). Los resultados obtenidos muestran reducciones de carácter exponencial del coeficiente de escorrentía y de la concentración de sedimentos asociados a la cubierta de la vegetación. Simultáneamente, se ha observado un retraso general en el comienzo y estabilización de la escorrentía superficial, así como un aumento de la profundidad alcanzada por el frente de humectación. Las variaciones estacionales en la humedad y el estado de la superficie del suelo también condicionaron las respuestas hidrológicas obtenidas; no obstante, en el caso de la tasa de infiltración final estas variaciones fueron atenuadas por la cubierta vegetal. Finalmente se ha determinado un umbral de carácter práctico para el desarrollo y valoración de las restauraciones mineras aplicable en ambientes de características similares al estudiado: cubiertas vegetales superiores al 50% permiten obtener un control biológico óptimo de las respuestas hidrológicas del suelo.

Palabras clave: costras superficiales, erosión, escorrentía, lluvia simulada, restauración, variaciones estacionales.

Effect of vegetation cover on the hydrology of reclaimed mining soils under Mediterranean-Continental climate

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Abstract

Vegetation cover plays a major role in the restoration and stabilization of disturbed systems. The analysis of relationships between restored vegetation and soil hydrology has special relevance for the evaluation and operation of mining reclamation, particularly in Mediterranean-Continental environments, where climatic conditions restrict the development of continuous vegetation cover. The effect of herbaceous vegetation cover on soil hydrology was analysed by means of rainfall simulation (63 mm h^{-1}; 0.24 m^2) in reclaimed soils derived from opencast coal mining (a non-saline and clay-loam textured spoil) in central-eastern Spain. A total of 75 simulation experiments were conducted at three different times throughout the year (late winter, summer and autumn) to control the influence of seasonal climatic fluctuations. Sediment concentrations in runoff and the runoff coefficient decreased exponentially with vegetation cover, while increases in steady infiltration rates were obtained with vegetation cover. Additional delays in runoff responses (longer time to runoff start and stabilization) and increases in the wetting front depth were observed with vegetation cover. Seasonal variations in soil surface state and moisture strongly influenced hydrological responses; although the influence of season on the analysed hydrological responses was attenuated by vegetation cover, especially in the case of infiltration rates. We also determined a practical ground cover threshold for site restoration and evaluation of over 50% vegetation cover, which could help achieve an optimum biological control of hydrological soil responses in the studied environment.

Key words: crust; erosion; restoration; runoff; seasonal variations; simulated rainfall.

Introduction

Reclaimed land from opencast mining activities is particularly sensitive to degradation by accelerated soil erosion (Nicolau and Asensio, 2000; Moreno-de las Heras *et al*., 2008), especially during the initial stages of site rehabilitation (Loch, 2000). In fact, freshly reclaimed mining soils commonly show unbalanced hydrologic behaviours characterized by a low infiltration capacity and high soil erodibility (Ritter, 1992; Guebert and Gardner, 2001; Ward *et al*., 1983). This is particularly important when topsoil replacement is not included in the rehabilitation procedure. These characteristics are frequently linked with the generation of important amounts of overland flow, leading to the acceleration of soil erosion processes, with dramatic consequences for on-site rehabilitation and off-site ecosystem damage (Nicolau, 2003).

Encouraging early vegetation establishment is essential in order to reduce the risk of degradation in these artificial systems (Wali, 1999; Whisenant, 2005). Important changes in the hydrological behaviour of reclaimed mining soils are usually driven by vegetation, which increases infiltration rates and prevents soil erosion (Sanchez and Wood, 1989; Loch, 2000). Indeed, vegetation protects the soil surface against the impact of raindrops, reduces the energy of runoff and stimulates the formation and stabilization of soil aggregates (Bochet *et al*., 1999; Durán-Zuazo and Rodríguez-Plequezuelo, 2008).

There is evidence that the interaction between vegetation and soil hydrology is generally nonlinear and often, this is conditioned by sharp thresholds in vegetation cover (Thornes, 2004). As a result two contrasted hydrological behaviours have been suggested for reclaimed mining soils: one characterised by the prevalence of biological controls, where plants actively mediate soil responses; and another characterised by the prevalence of abiotic controls, where crusting and rilling processes control runoff generation and soil erosion (Nicolau and Asensio, 2000). In the latter case, soil hydrology is strongly influenced by seasonal climatic fluctuations (Nicolau, 2002; Martínez-Murillo and Ruiz-Sinoga, 2007). In fact, seasonal variations in soil surface state and soil moisture can lead to large shifts in soil hydrological responses under Mediterranean climate conditions (Cerdà, 1997; Regüés and Gallart, 2004).

Vegetation cover is considered a key indicator of restoration success, as it can reflect critical stages of ecosystem development and functionality (Vallauri *et al*., 2005). The determination of optimum vegetative cover thresholds which ensure the biological control of hydrological processes has been stressed as an important goal for the restoration of both natural and man-made landscapes (Snelder and Bryan, 1995; Gutierrez and Hernandez, 1996; Andrés and Jorba, 2000). This is particularly important in areas where climatic restrictions severely constrain the development of a continuous vegetation cover, as in Mediterranean-Continental environments.

The aim of this investigation was to study the influence of the herbaceous vegetation cover on the hydrological response of reclaimed mining soils by means of simulated rainfall, in a Mediterranean-Continental environment (central-eastern Spain). Experiments were carried out at different times throughout the year to assess the seasonal influences on soil hydrology. As a practical objective, we aimed to identify an optimal threshold of vegetation cover to achieve satisfactory biological control of soil hydrology in the studied environment.

Site description

This work was carried out in the Utrillas field site, which is located in the reclaimed mine El Moral (Utrillas coalfield), central-eastern Spain (40º47'24'' N, 0º47'24'' W; 1100 m.a.s.l.; Fig. 4.1a). The climate is Mediterranean-Continental. Mean annual air temperature is 11 ºC (6.8 ºC in December and 23.5 ºC in July). The local moisture regime is Mediterranean-dry, according to Papadakis (1966). Mean annual precipitation is 466 mm (concentrated in spring and autumn) and potential evapotranspiration is 758 mm. Vegetation development in this area is constrained by a long frost period (from October to April) and an intense summer drought (from June to October).

The study site consists of five reclaimed slopes located in the north-facing side of a spoil-bank (Fig. 4.1b). These slopes were restored using very similar treatments during 1988-89 by the Minas y Ferrocarril de Utrillas S.A. company. Slope angle is 20º; the substrate used to cover the spoil-bank is overburden material from the Escucha cretacic formation, of Albian age. This is a non-saline and clay-loam textured spoil (kaolinitic-illitic mineralogy) with basic pH (Table 4.1). Revegetation was undertaken by sowing a mixture of perennial grasses and leguminous herbs (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis*, *Lolium perenne*, *Medicago sativa* and *Onobrychis viciifolia*; nomenclature follows Tutin *et al*., 1964-1980).

The experimental slopes differ in their geomorphological design. A flat and bare area (slope 4-6º, 6-9 m long) is directly connected to the top of two slopes (slopes 1 and 2). Similarly, a bare steep bank (slope 40º, 3-7 m long) is connected to the top of another two slopes (slopes 3 and 4). The last slope (slope 5) has no connected structures. These up-slope structures work as water contributing areas, generating overland flow that promotes soil erosion processes. Eighteen years of soil erosion processes of different intensity (historical rill erosion rate from 0 t ha^{-1} $year^{-1}$ in slope 5 up to about

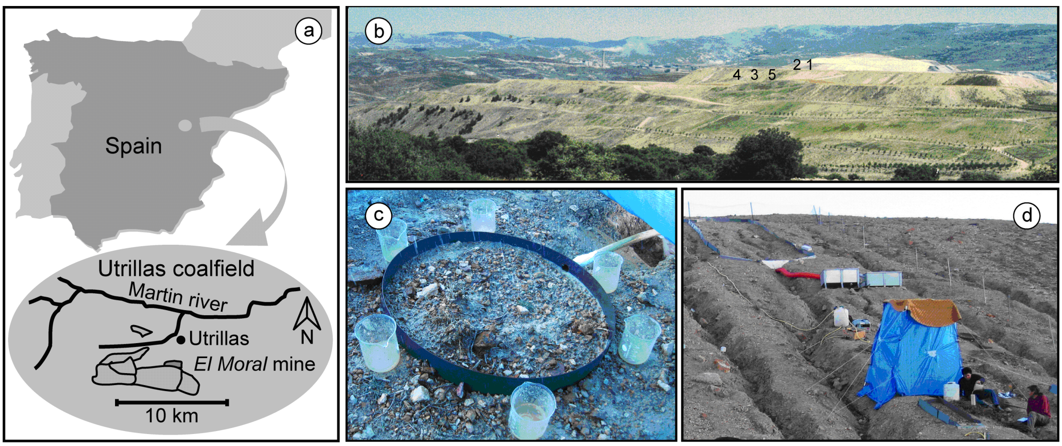

Figure 4.1. Site diagram: (a) location map; (b) frontal view of the "El Moral" spoil-bank with the situation of the five experimental slopes (numbers 1-5); (c) outlook of one permanent plot during the operation of a rainfall simulation experiment; (d) detail of the rainfall simulation setup and a 3 x 15-m (wide x length) bounded plot in slope 1.

70 t ha^{-1} year^{-1} in slope 1; Table 4.1) have induced large differences in vegetation development (from about 1% total vegetation cover in slope 1 up to 60% in slope 5) and some associated soil traits (soil organic matter, total nitrogen and bulk density) in these five slopes, following a degradation gradient. A general description of soil, and vegetation traits as well as erosion features of the five experimental slopes is presented in Table 4.1.

Methods

Rainfall simulation setup

Experiments were carried out using a single nozzle (HARDI® 1553) rainfall simulator based on the model described by Cerdà *et al*. (1997). Rainfall simulations were performed using a pressure of 1.5 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 63.4 mm h^{-1}; rainfall uniformity (*sensu* Christiansen, 1942) was 81.1 %; 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}. Natural rainfall of similar intensity during 30-60 min (range used for rainfall simulations) has a return period of 5-15 years in central-eastern Spain (Elías and Ruíz, 1979).

Twenty-five steel rings (55 cm diameter, 15 cm

Table 4.1. Basic characteristics (Mean ± S.D.) of the five experimental slopes.

	N	Slope 1	Slope 2	Slope 3	Slope 4	Slope 5
Date of reclamation		1989	1989	1988	1988	1988
Topography						
Slope length (m)		55	50	75	75	60
Slope gradient (º)		20	20	20	20	20
Water-contributing area (m)		8.0	8.0	6.5	4.0	0.0
Aspect		North	North	North	North	North
Soil traits						
Stoniness [1] (%)	25	22.2 ± 2.2	24.7 ± 3.5	26.2 ± 4.1	25.2 ± 2.6	24.5 ± 3.3
Sand [1] (%)	25	33.6 ± 3.6	33.5 ± 3.7	33.8 ± 3.0	39.9 ± 1.8	36.3 ± 2.7
Silt [1] (%)	25	26.9 ± 2.8	33.8 ± 1.6	30.8 ± 1.8	26.4 ± 2.9	26.6 ± 4.5
Clay [1] (%)	25	39.5 ± 2.2	32.8 ± 2.9	35.4 ± 2.1	33.8 ± 2.1	37.1 ± 2.9
Texture	25	Clay loam	Clay loam	Clay loam	Clay loam	Clay loam
EC [1] -w/v: ½- (dS m^{-1})	25	0.24 ± 0.13	0.26 ± 0.14	0.20 ± 0.10	0.19 ± 0.03	0.23 ± 0.03
pH [1] -H_2O; w/v: ½-	25	8.03 ± 0.12	7.96 ± 0.14	7.95 ± 0.13	7.95 ± 0.13	7.91 ± 0.10
Organic matter [1] (%)	25	0.58 ± 0.20 b	0.56 ± 0.23 b	1.27 ± 0.35 ab	1.46 ± 0.83 ab	2.00 ± 0.74 a
Total nitrogen [1] (%)	25	0.04 ± 0.01 b	0.03 ± 0.01 b	0.07 ± 0.02 ab	0.07 ± 0.04 ab	0.10 ± 0.04 a
Bulk density [2] (g cm^{-3})	75	1.51 ± 0.14 a	1.49 ± 0.12 a	1.39 ± 0.17 a	1.39 ± 0.12 a	1.23 ± 0.17 b
Vegetation traits						
Vegetation cover [3] (%)	150	1.1 ± 2.0 c	8.2 ± 5.5 c	27.8 ± 9.9 b	44.3 ± 16.2 ab	59.4 ± 20.8 a
Erosion features						
Rill density [4] (m m^{-2})	15	0.95	0.78	0.58	0.30	0.0
Rill erosion rate [5] (t ha^{-1}year^{-1})	5	71.41	45.03	16.95	7.86	0.00

N: number of samples; EC: electrical conductivity; w/v: relation weight (soil) / volume (water).
[1] Measured in composite samples (each formed by six subsamples) from the first 10 cm, randomly distributed.
[2] Measured in unaltered soil cores (3 cm height by 5 cm diameter) randomly distributed.
[3] Green vegetation plus litter visually estimated in randomly distributed 0.25 m^2 plots during spring 2006.
[4] Linear rill length (m) per surface area (m^2).
[5] Measured from rill network dimensions following Morgan (1997).
Values with the same letters (a-c) within rows do not differ significantly at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney.

height) were installed (by inserting them 5 cm into the soil) at inter-rill areas on the five experimental slopes (5 plots per slope). Each of these 0.24 m^2 permanent plots has a drainpipe outlet (2.5 cm diameter) to collect runoff and sediments (Fig. 4.1c). The rings were installed in October 2005, five months before the first experiment, in order to ensure the natural consolidation of soil surface disruptions caused during the set up.

Sampling

During 2006, 75 rainfall experiments were conducted throughout three rainfall simulation campaigns (5 rainfall events per slope and campaign), performed at different times: March (transition between winter and spring), August (summer) and October (autumn). These three times were selected to study the most representative and contrasted soil surface states: disrupted (March), dry and crusted (August) and finally, moist and crusted (October).

The duration of each rainfall event was variable (until stabilization of runoff rate, at least 30 min). Runoff discharge in each rainfall experiment was collected manually at 1 min intervals (from the start of runoff) with the aid of two small jars of 0.5 l capacity (Fig. 4.1d); the volume of each 1-min runoff sample was immediately measured with a graduated cylinder and transferred to a plastic bucket (20 l capacity). After each rainfall experiment, runoff samples stored in the bucket were mixed, and a homogeneous aliquot (0.5 l) was extracted. Sediment concentration (*Sc*, g l^{-1}) in aliquots was determined by weighing after oven drying (105 ºC). Time to runoff (*Tr*, min) and runoff coefficient (*Qc*, %) were also measured. Finally, after each simulation, a vertical cut in the soil profile (outside but close to simulation plots) was done; the depth reached by the wetting front was measured (using a tape measure) as the thickness of the moistened section in the exposed soil profile (cm). This depth was divided by the duration of rainfall experiments (which was different in each experiment) to obtain a comparable value: the soil profile moistening rate (*Mr*, cm h^{-1}), which describes the downward velocity of the wetting front in the soil profile. To minimize the influence of water salinity on hydrological soil responses (Agassi *et al*., 1981), good quality water (EC= 0.28 dS m^{-1}; SAR <0.01) was used.

In each plot and before each experimental campaign, the cover (%) of green vegetation and litter (mainly standing dead vegetation) was visually estimated. A variable that integrated both green vegetation and litter was used for analyses (vegetation cover, %). This approximation has been widely adopted for the analysis of the relationship between vegetation and hydrological processes, as dead vegetation can also intercept rainfall and offer resistance to runoff (Elwell and Stocking, 1976; Francis and Thornes, 1990; Loch, 2000).

Soil surface dynamics (soil surface strength and soil moisture) was monitored from October 2005 to October 2006. These measurements were periodically taken outside the experimental plots to avoid the alteration of soil surface conditions. In order to assess the condition of soil surface crusts, measurements of soil surface strength, measured as penetration resistance (kg), were taken using a pocket penetrometer with a 0.64 cm diameter flat probe (Geotester®). Every two months, twenty random measurements of soil surface strength were taken at inter-rill areas in each slope. Measurements of volumetric soil moisture (%) were obtained from a network of TDR sensors (formed by eight randomly distributed sensors placed at inter-rill areas in each slope), which monitored the first 15 cm of the soil profile. We followed the methodology proposed by Cassel *et al*. (1994), using a TDR instrument (Tektronix® 1502C) which provided an accuracy of 94% in the determination of soil moisture. Soil moisture measurements were taken periodically (every 15 days without rain and 24 h after each natural rainfall event).

Complementary information on annual cumulative sediment yield measured at a larger scale (including active rilling processes) under natural rainfall conditions is provided to discuss the scale-related limitations of the results reported in this work. This data was obtained in 3 x 15-m (wide x length) plots (Fig. 4.1d) placed in the analysed experimental slopes (one plot per slope) from October 2005 to October 2006 (Moreno-de las Heras *et al*., 2007).

Data analyses

Infiltration was estimated as the balance of precipitation (intensity of rainfall simulations) and minute-measured runoff. Two descriptor parameters of the infiltration phenomenon (the final infiltration rate-I_f and the shape coefficient of the apparent infiltration curve-K) were obtained by fitting, from minute-estimated infiltration values, the Horton-type equation proposed by Borselli *et al*. (1996):

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

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

I_f and K parameters, time to runoff (Tr), runoff coefficient (Qc), soil profile moistening rate (Mr) and sediment concentration (Sc) have been used as descriptors of the soil hydrological response to simulated rainfall. In this way, I_f, Qc, Mr and Sc describe the magnitude of responses, while Tr and K describe the speed of responses (greater Tr and K values mean a longer time to the start of runoff and to the stabilization of the infiltration rate).

Principal component analysis (PCA) was performed introducing all parameters (I_f, Qc, Mr, Sc, Tr, K) to express the main gradients. The relative influence of each rainfall simulation response parameter on the gradients extracted by PCA was analysed using the Spearman´s R correlation coefficient. The impact of vegetation cover was analysed by fitting a second-order polynomial surface to the spatial configuration of PCA, using general additive models and interpolating the fitted values on a PCA biplot.

Differences in response parameters (I_f, Qc, Mr, Sc, Tr, K) between the five experimental slopes were determined using Kruskal-Wallis analysis of variance and Mann-Whitney *post-hoc* tests. Within each slope, differences between experimental campaigns were analysed using the Friedman analysis for repeated measurements and Wilcoxon *post hoc* tests for paired groups.

To explain the links between vegetation and hydrological responses, the best fitting regression function between each hydrological response parameter (I_f, Qc, Mr, Sc, Tr, K) and vegetation cover was calculated.

In order to establish an optimum ground cover threshold for the influence of vegetation on the hydrological response of soils, a sequence of one-way non-parametric multivariate analyses of variance (hereafter NPMANOVA; Anderson, 2001) were carried out. All of these tests were performed using all hydrological response parameters as dependent variables. Several discrete variables were used as factors in each test. These variables were obtained by splitting vegetation cover into different cover categories (1; 5, 10; 15; …; 80; 85; 90%), and thus creating a factor for each category. Thus, a total of 19 factors were obtained, each with two levels, where a value of 1 is assigned to all cases with over X% cover and 0 is assigned to all cases with under X% cover (X = the appropriate percentage in each factor). NPMANOVA tests were carried out for each of these factors, obtaining a sequence of F-statistics. A diagram plotting each F-statistic *versus* the vegetation cover category of each factor was plotted to find the range of covers which maximised the F-statistic and, in consequence, maximised differences in the overall hydrological response between groups with different vegetation cover. This method constitutes a new approach for determining critical thresholds using a statistical procedure, avoiding the subjective interpretation of multiple link functions.

Finally, in order to analyse the possible controlling effect of vegetation on inter-seasonal changes of soil hydrological responses, the variation coefficients of all studied response parameters (I_f, Qc, Mr, Sc, Tr, K) were calculated between the three rainfall simulation campaigns, for each experimental plot. The relationships between mean annual vegetation cover and the coefficients of variation of each parameter were explored using Spearman´s correlations.

PC-ORD ver. 4 (McCune and Mefford, 1999) was used for PCA. The ordisurf function of the Vegan package from the R system (Oksanen *et al*., 2007) was used to fit the second-order polynomial surface of vegetation cover to the PCA biplot. NPMANOVA and other statistics were performed using the PAST ver. 1.67b package (Hammer *et al*., 2001).

Results

Seasonal changes in soil surface state and soil moisture

Both soil moisture and soil surface state showed large seasonal variations (Fig. 4.2):

- Autumn and winter precipitation (295.74 mm) increased soil moisture from October 2005 (11-17%) to March 2006 (19-23%). An important decrease in soil moisture, from 20% to almost 10%, occurred between March and June 2006, when total precipitation was only 70.75 mm. Several high intensity convective thunderstorms (I_{30} up to 40 mm h^{-1}) produced 165.6 mm of precipitation, which increased soil moisture sharply during late spring and early summer, with values up to 17-20%. Soil moisture quickly decreased to near 9% at the end of summer. The first autumn rains (83.34 mm) increased soil moisture again, reaching 15-20%.

- Coinciding with the beginning of the frost period, soil surface strength gradually decreased during autumn and winter, reaching minimum resistance to penetration in March 2006. Between the end of winter and the beginning of spring, the soil surface was highly disrupted, and had a spongy morphology. Subsequent increases in soil strength, of different intensity, took place until summer, when the highest values of soil strength were reached and a

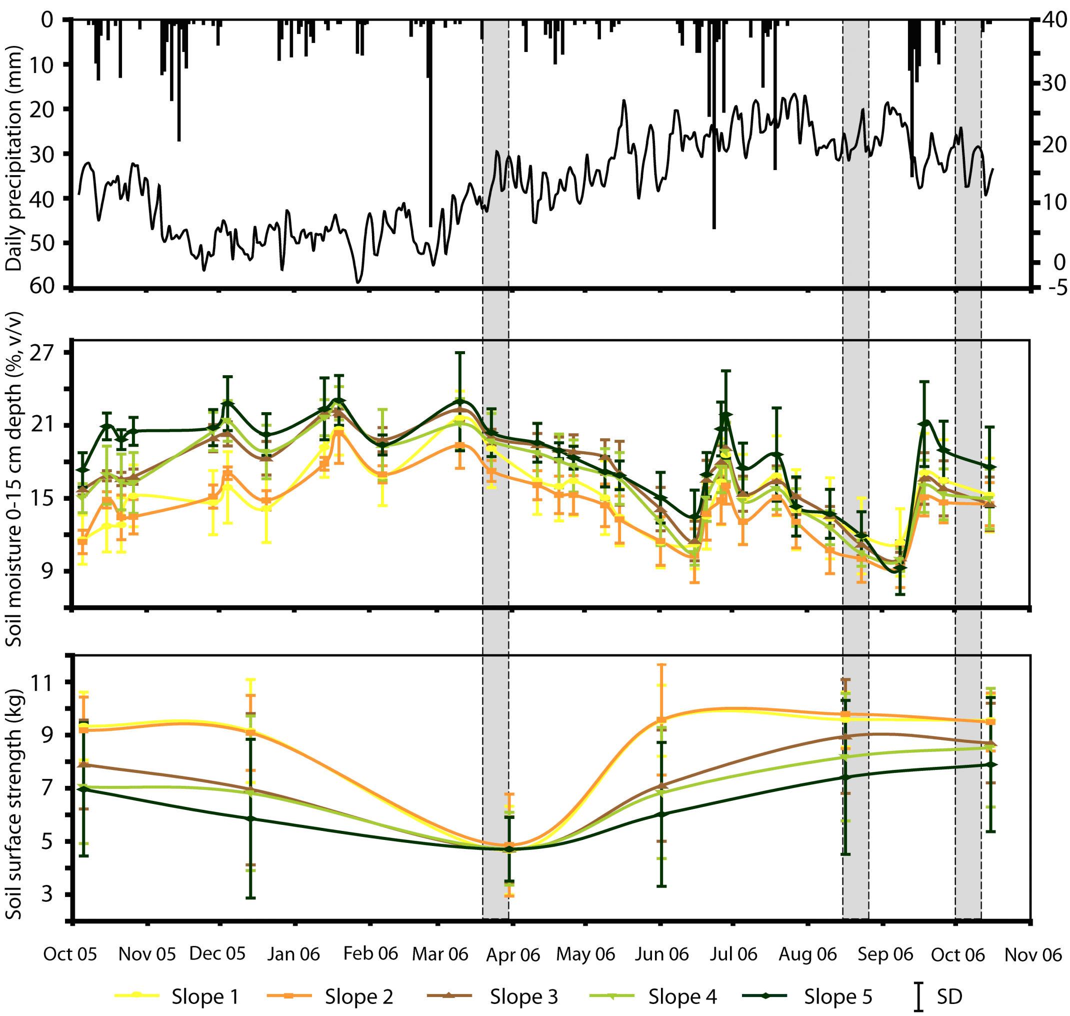

Figure 4.2. Meteorological data (daily precipitation and daily mean air temperature), soil moisture and soil surface strength in the five experimental slopes from October 2005 to October 2006. Vertical grey strips show the dates of the three rainfall simulation campaigns.

dense soil crust developed in bare surfaces. There were some important differences in soil surface dynamics between the studied slopes: the decrease in soil surface strength registered during autumn and winter was delayed in slopes 1 and 2 (the least vegetated) until the beginning of winter, when frosts were more recurrent and severe. Likewise, the increase of soil surface strength in these slopes was faster than in the most vegetated ones (slopes 3, 4 and 5), wherein the maximum development of soil crusts was not achieved until the occurrence of high intensity convective storms during late spring and early summer.

Analysis of response gradients

The first two dimensions extracted from the PCA explained 79.19% of the variance (component 1: 62.33% and component 2: 16.86%).

Component 1 was strongly and negatively correlated with I_f, *Mr*, *Tr* and *K* and positively with *Qc* (Table 4.2). Thus, this component represents a hydrological gradient, opposing long response times and high infiltration rates to short response times and high runoff rates (Fig. 4.3). Component 2 was strongly and negatively correlated with *Sc* (Table 4.2). Therefore this component is related to soil erodibility, representing decreases in sediment yield (Fig. 4.3).

Hydrological soil responses to simulated rainfall were distributed within the PCA configuration along the gradient of vegetation cover inherent to the experimental slopes (Fig. 4.3). In fact, a second-order polynomial surface, which represents vegetation cover, was successfully fitted onto the spatial configuration of PCA (R^2=0.84; p<0.000). According to this, increases in vegetation cover (negatively associated with component 1 and positively associated with component 2) can be linked to increases in infiltration and response time, and reductions in sediment concentrations.

Table 4.2. Correlation coefficients (Spearman's R) between the first two components extracted by PCA and rainfall simulation response parameters.

	Spearman's R coefficient	
	Component 1	Component 2
Tr	**-0.649*****	-0.029
K	**-0.605*****	-0.177*
I_f	**-0.775*****	0.142
Qc	**0.888*****	-0.046
Mr	**-0.685*****	-0.084
Sc	0.260***	**-0.596*****

Abbreviations: *Tr*: time to runoff; *K*: shape coefficient of infiltration function; I_f: final infiltration rate; *Sc*: sediment concentration.
Coefficients in bold are >0.500.
* Significant at α=0.05; *** Significant at α=0.001.

Another trend linked to inter-seasonal changes was observed: a general increase in runoff as well as a decrease in response time and sediment concentrations can be deduced from

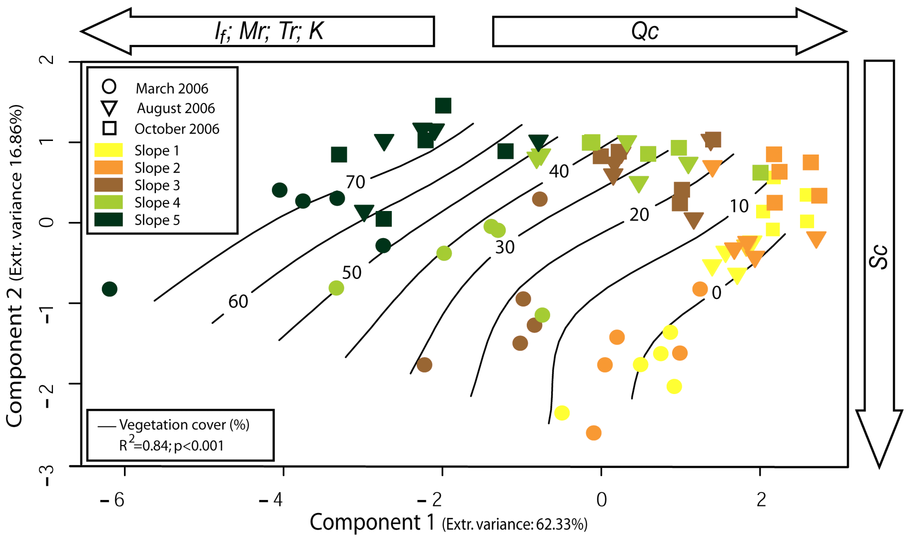

Figure 4.3. Diagram representing the two first components extracted by PCA. Rainfall simulations from the five experimental slopes are represented by different colours. The three different experimental campaigns are represented by different symbols. Vegetation cover is represented by the contours derived from a fitted polynomial surface. External arrows indicate the correlation direction of significant response parameters (I_f: final infiltration rate; *Mr*: soil profile moistening rate; *Tr*: time to runoff; *K*: shape coefficient of infiltration function; *Qc*: runoff coefficient; *Sc*: sediment concentration).

variations of case positions in the PCA biplot, from the March experiments to the August and October experiments (oblique to both components; Fig. 4.3).

Differences between slopes and inter-seasonal variations

In agreement with gradient analysis, important differences in response parameters (*Tr*, *K*, I_f, *Qc*, *Mr* and *Sc*) to rainfall simulations were found between experimental slopes and campaigns (Table 4.3).

Following the degradation gradient from slope 5 to slopes 1 and 2, there were large differences in mean response times to rainfall simulations. Slow responses found in slope 5 (*Tr*~ 6 min, and *K*~ 9) contrasted with fast responses in the least vegetated slopes (*Tr*~ 1.4 min, and *K*~ 4 in slopes 1 and 2; Table 4.3). Values of variables describing the magnitude of responses were also very different. In this way, small runoff coefficients and high, steady infiltration and soil profile moistening rates were obtained in the most vegetated slope (*Qc*~ 30 %, I_f~ 37 mm h^{-1}, and *Mr*~ 24 cm h^{-1} in slope 5; Table 4.3). On the other hand, high runoff coefficients and low infiltration and soil profile moistening rates were obtained in the least vegetated slopes (*Qc*~ 70 %, I_f~ 10 mm h^{-1}, and *Mr*~ 8 cm h^{-1} in slopes 1 and 2; Table 4.3). Similarly, low sediment concentrations in slope 5 (*Sc*~ 2 g l^{-1}) contrasted with greater concentrations in slopes 1 and 2 (*Sc*~ 12 g l^{-1}; Table 4.3).

In general, all responses were affected by inter-seasonal variations. In this way, time needed to initiate runoff and reach a steady infiltration rate (described by *Tr* and *K* parameters) was in general lower in summer and autumn (August and October experiments) than during the transition between winter and spring (March experiments; Table 4.3). However, these differences were not significant in the case of slope 5. Similar variations were identified for *Mr* in all slopes, highlighting reductions in the rate at which the wetting front descends along the soil profile, from values attained during the transition between winter and spring to those reached in summer and autumn (Table 4.3). Final infiltration rates (I_f) and runoff coefficients (*Qc*) revealed opposite seasonal trends: I_f showed a gradual decrease from March 2006 values to those from October 2006. On the other hand, *Qc* showed a progressive increase from the transition between winter and spring to autumn (Table 4.3). Although these variations were significant for all slopes, the magnitude of change was different (especially in the case of I_f) depending on the slope considered. Indeed, I_f decreased over 60% in the most degraded slopes (slopes 1 and 2) from March to October 2006 (from *circa* 15 to *c*. 5 mm h^{-1}), while this reduction was approximately 10% (from *c*. 40 to *c*. 35 mm h^{-1}; Table 4.3) in the most vegetated slope (slope 5). Measurements of sediment concentrations (*Sc*) also varied throughout the year, decreasing progressively from March to October 2006. These variations were similar in all slopes, with a three-fold decrease in most cases (Table 4.3).

Influence of vegetation cover

The relationship between vegetation cover and hydrological response parameters was different in each case (Fig. 4.4). This relationship was best described by exponential decay functions in the case of sediment concentration (*Sc*: R^2=0.61; p<0.000) and runoff rate (*Qc*: R^2=0.59; p<0.000). Linear fits worked best for the final infiltration rate (I_f: R^2=0.64; p<0.000)

Table 4.3. Differences in response parameters after rainfall simulations (Mean ± S.D.) at the different experimental slopes during all campaigns.

		Tr- Time to runoff (min)	*K*- Shape coefficient	I_f- Final infiltration rate (mm h^{-1})	*Qc*- Runoff coefficient (%)	*Mr*- Soil profile moistening rate (cm h^{-1})	*Sc*- Sediment concentration (g l^{-1})
Slope 1	March	1.92 ± 0.61 a	6.94 ± 3.40 a	16.82 ± 2.92 a	57.27 ± 6.75 c	15.05 ± 2.20 a	19.47 ± 0.70 a
	August	1.04 ± 0.31 b	2.62 ± 0.41 b	13.84 ± 2.10 b	70.10 ± 3.23 b	6.06 ± 0.79 b	13.29 ± 1.21 b
	October	1.00 ± 0.46 b	2.70 ± 0.97 b	3.69 ± 2.78 c	84.53 ± 4.60 a	5.24 ± 1.36 b	8.19 ± 1.57 c
	Mean	*1.32 ± 0.16* B	*4.18 ± 1.53* AB	*11.45 ± 2.19* B	*70.63 ± 4.43* A	*8.79 ± 1.31* B	*13.65 ± 0.52* A
Slope 2	March	2.75 ± 1.51 a	6.96 ± 3.62 a	14.08 ± 4.30 a	56.98 ± 8.13 c	13.47 ± 2.25 a	18.15 ± 1.92 a
	August	1.00 ± 0.34 b	1.94 ± 0.30 b	11.32 ± 5.11 b	75.65 ± 8.40 b	6.46 ± 1.06 b	11.24 ± 3.25 b
	October	0.80 ± 0.20 b	1.96 ± 0.25 b	4.93 ± 3.91 c	85.55 ± 5.34 a	3.75 ± 0.29 c	6.40 ± 1.85 c
	Mean	*1.51 ± 0.51* B	*3.62 ± 1.19* B	*10.11 ± 3.60* B	*72.73 ± 5.47* A	*7.89 ± 1.00* B	*11.93 ± 2.15* A
Slope 3	March	2.57 ± 0.64 a	8.05 ± 4.58 a	26.69 ± 2.03 a	43.05 ± 4.70 c	27.38 ± 1.14 a	12.28 ± 4.17 a
	August	1.97 ± 0.64 a	2.94 ± 0.34 b	19.79 ± 5.28 b	59.50 ± 9.36 b	9.05 ± 2.62 b	5.61 ± 2.64 b
	October	1.82 ± 0.84 a	3.44 ± 1.19 b	14.97 ± 9.03 c	64.76 ± 9.83 a	12.92 ± 1.91 b	4.18 ± 1.30 c
	Mean	*2.12 ± 0.67* AB	*4.81 ± 1.43* AB	*20.48 ± 4.61* AB	*55.57 ± 7.33* AB	*16.45 ± 1.62* AB	*7.36 ± 2.56* AB
Slope 4	March	2.85 ± 0.75 a	8.22 ± 4.31 a	27.53 ± 6.25 a	42.98 ± 9.56 c	35.49 ± 5.12 a	7.28 ± 4.24 a
	August	1.53 ± 0.29 b	3.97 ± 1.65 a	23.65 ± 9.97 ab	53.22 ± 12.85 b	13.45 ± 5.18 b	3.76 ± 1.79 b
	October	0.94 ± 0.53 b	4.47 ± 0.73 a	16.86 ± 9.59 b	63.36 ± 14.78 a	11.76 ± 1.58 b	2.37 ± 1.03 c
	Mean	*1.77 ± 0.21* AB	*5.55 ± 1.86* AB	*22.68 ± 8.12* AB	*53.19 ± 11.78* AB	*20.23 ± 2.32* A	*4.47 ± 2.29* AB
Slope 5	March	7.35 ± 3.72 a	11.04 ± 5.43 a	39.14 ± 6.12 a	24.22 ± 8.78 b	38.42 ± 7.15 a	2.76 ± 0.62 a
	August	5.77 ± 2.49 a	8.18 ± 4.58 a	36.18 ± 7.23 ab	31.95 ± 9.14 a	12.55 ± 1.24 b	1.36 ± 0.21 b
	October	5.45 ± 2.90 a	7.52 ± 5.31 a	34.63 ± 7.85 b	31.15 ± 8.29 a	19.48 ± 3.62 b	1.06 ± 0.11 c
	Mean	*6.19 ± 2.91* A	*8.91 ± 3.95* A	*36.65 ± 7.01* A	*29.11 ± 8.46* B	*23.47 ± 3.49* A	*1.73 ± 0.29* B

Considering simulation times in each slope, data followed by different small letters (a-c) are significantly different at α=0.05. Tested using Friedman ANOVA and *post-hoc* Wilcoxon tests.
Considering experimental slopes, mean values followed by different capital letters (A-B) are significantly different at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whithney.

and the shape coefficient of infiltration function (*K*: R^2=0.17; *p*<0.000). Other polynomial functions best described the relationship between vegetation cover and time to runoff (*Tr*: R^2=0.48; *p*<0.000) and the soil profile moistening rate (*Mr*: R^2=0.25; *p*<0.000).

Based on the diagram which represents F-sta-

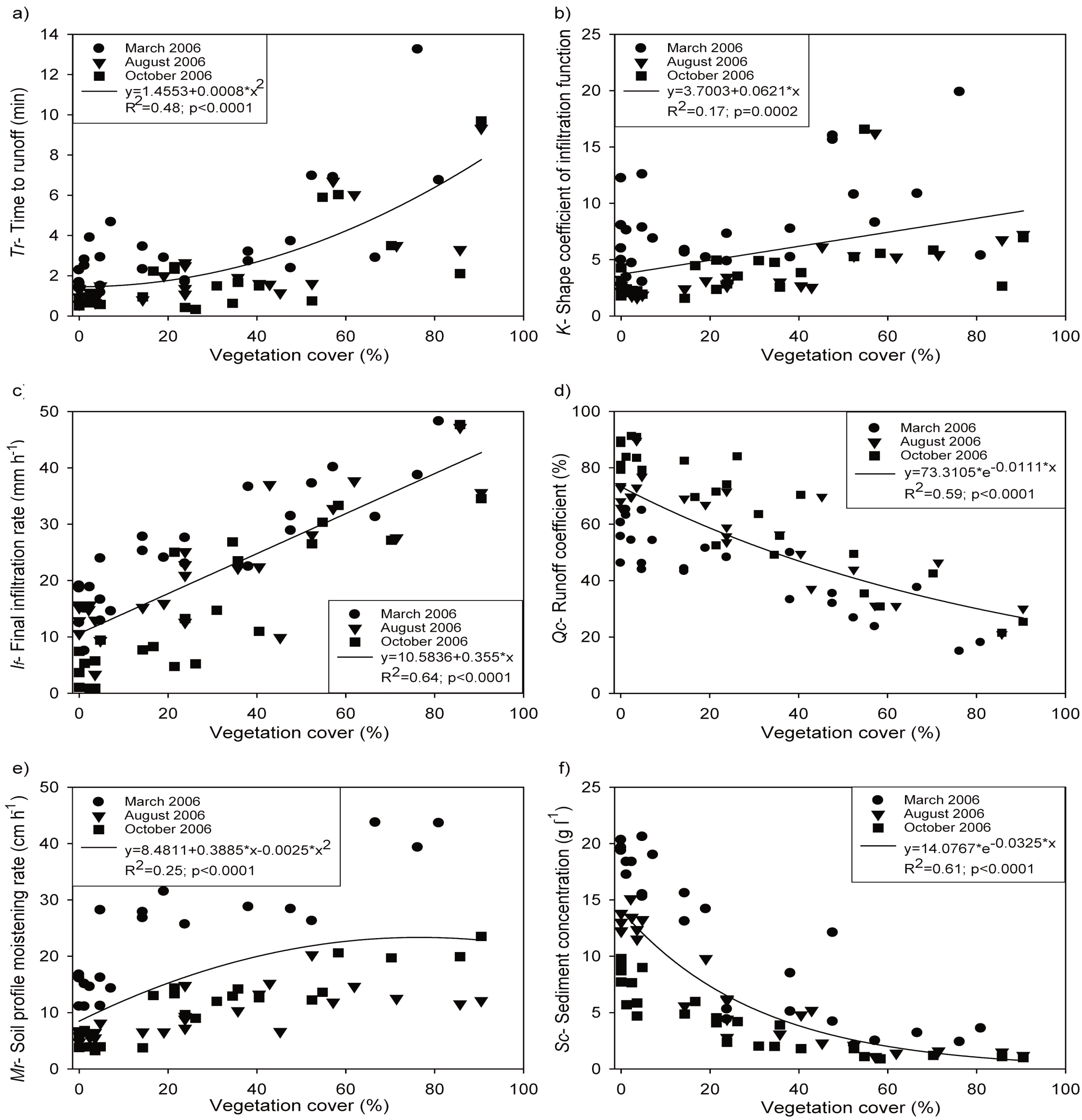

Figure 4.4. Diagram representing the two first components extracted by PCA. Rainfall simulations from the five experimental slopes are represented by different colours. The three different experimental campaigns are represented by different symbols. Vegetation cover is represented by the contours derived from a fitted polynomial surface. External arrows indicate the correlation direction of significant response parameters (I_f: final infiltration rate; *Mr*: soil profile moistening rate; *Tr*: time to runoff; *K*: shape coefficient of infiltration function; *Qc*: runoff coefficient; *Sc*: sediment concentration).

tistics of the NPMANOVA tests, performed with all the response parameters (*Tr*, *K*, I_f, *Qc*, *Mr* and *Sc*) as dependent variables and each vegetation cover category as independent factors (Fig. 4.5), the F-statistic is maximum at 30-50% vegetation cover. This means that the hydrological response of cases with less than 30% vegetation cover is very different from that of cases with cover above 50%.

For those parameters which are most strongly linked with vegetation cover (I_f, *Qc* and *Sc*), Figure 4.4 shows a large dispersal range of values in cases with under 30% cover, while cases with over 50% cover show limited dispersion. However, if only inter-seasonal variability is considered, just one significant effect of vegetation cover on response parameters was identified (Table 4.4). This effect was a strong negative link between vegetation cover and the inter-seasonal coefficient of variance of the final infiltration rate. This indicated lower seasonal variability in steady infiltration in the most vegetated plots when compared to the least vegetated plots.

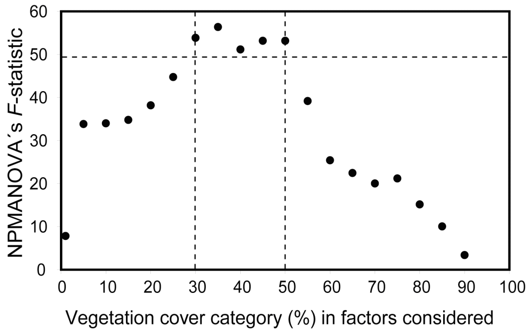

Figure 4.5. Diagram representing the F-statistics obtained for each vegetation cover category considered in the sequence of non-parametric multivariate analysis of variance (NPMANOVA) performed with the hydrological response parameters. Vertical doted lines show the range of covers which maximise differences in global hydrological response between plots of different vegetation cover.

Discussion

The general trend extracted from gradient analysis shows a strong link between vegetation and soil hydrological responses to simulated rainfall (Fig. 4.3). This agrees with several studies, which emphasize the positive influence of vegetation on the hydrological behaviour of both natural (Castillo *et al*., 1997; Marqués *et al*., 2007) and reclaimed soils (Sanchez and Wood, 1989; Loch, 2000; Cerdà, 2007). Furthermore, vegetation development largely explained the differences in soil hydrological responses between experimental slopes. In fact, vegetation can induce important changes in the hydrological behaviour of soils, improving soil infiltration capacity and reducing soil erodibility (Elwell and Stocking, 1976; Bochet *et al*., 1999). Different relationships described the links between the analysed response para-

Table 4.4. Correlation coefficients (Spearman's R) between mean annual vegetation cover and the inter-seasonal coefficient of variation (VC) of each rainfall simulation response parameter.

	Spearman's R coefficient
	Averaged vegetation cover
Inter-seasonal VC of Time to runoff	-0.259
Inter-seasonal VC of Shape coefficient of infiltration function	-0.328
Inter-seasonal VC of Final infiltration rate	**-0.748*****
Inter-seasonal VC of Runoff coefficient	-0.195
Inter-seasonal VC of Soil profile moistening rate	-0.197
Inter-seasonal VC of Sediment concentration	-0.349

Coefficients in bold are >0.400.
*** Significant at α=0.001.

meters and vegetation cover (Fig. 4.4). A classic exponential decay function (Francis and Thornes, 1990) described a sharp reduction in sediment concentration with increasing vegetation cover. In accordance with previous studies (Pierson *et al*., 1994; Gimeno-García *et al*., 2007), the relationships between vegetation cover and both steady infiltration and runoff rates were also strong, but changes along the vegetation cover gradient were less drastic. Other response parameters showed weaker relationships with vegetation cover, which suggested a generalised delay of runoff responses (runoff start and stabilization) and an increase in the downward velocity of the wetting front in the soil profile.

As expected, the seasonal fluctuations of Mediterranean-Continental climate played a relevant role in the temporal variability of soil hydrological responses. In general, hydrological responses were slower during the transition between winter and spring and were characterised by higher infiltration rates and sediment concentrations (Table 4.3). This can be explained by the occurrence of freeze-thaw cycles from late autumn through the winter months, which disrupt the soil surface (Fig. 4.2), increasing soil porosity and erodibility. Similar findings have been reported in environments where freezing has a special relevance for soil hydrology, such as those under Mediterranean-Continental climate (Nicolau, 2002; Regüés and Gallart, 2004). Faster responses and subsequent decreases in infiltration rates and sediment concentrations took place throughout the year until autumn (Table 4.3). These can be explained by soil surface consolidation (associated to crusting), due to the occurrence of subsequent storms until summer, and a later increase in soil moisture at the beginning of autumn (Fig. 4.2). These variations in soil conditions (surface crusts and soil moisture) are widely recognised as essential sources of variability in soil hydrological responses (Le Bissonnais *et al*., 1998; Torri *et al*., 1999; Castillo *et al*., 2003).

In spite of intense seasonal fluctuations, vegetation cover had a strong influence on how response parameters were affected by seasonal changes. This is especially relevant in the case of the final infiltration rate (Table 4.3), as the high variations recorded in the least vegetated slopes (slopes 1 and 2) contrast with the low variations recorded in the most densely covered slope (slope 5). Indeed, vegetation cover had a direct influence on steady infiltration rates, reducing their temporal variability (Table 4.4). This is probably due to vegetation-induced changes of macropore soil structure and protection against surface crusting (Folley *et al*., 1991; Loch, 2000). In this way, the maximum development of surface crusts throughout the 2005-06 hydrologic year in slopes 1 and 2 was much higher than that of slope 5 (Fig. 4.2). This agrees with other findings that suggest a relevant role of vegetation in reducing the influence of seasonal climatic fluctuations on infiltration rates (Cerdà, 1997).

These results can be interpreted assuming the general non-linear nature of relationships between vegetation and hydrological processes (Thornes, 2004). In our case, the most dramatic differences were found between cases with under 30% and over 50% vegetation cover (Figure 4.5). According to the conceptual framework proposed by Nicolau and Asensio (2000), below this range (cover <30%), hydrological soil responses remain biologically uncontrolled (leading to important amounts of runoff and sediment yield) and are strongly dependent on

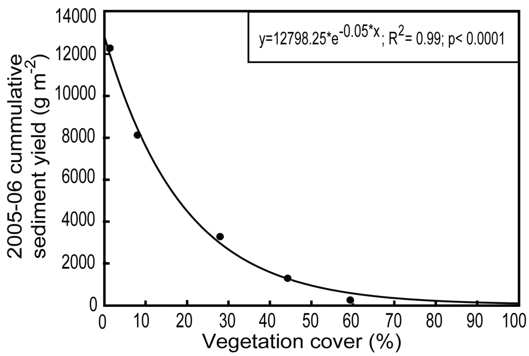

Figure 4.6. Relationship between vegetation cover and the annual cumulative sediment yield (from October 2005 to October 2006) recorded in bounded 3x15-m plots in the five experimental slopes (data from Moreno-de las Heras *et al*., 2007).

seasonal soil fluctuations (surface crusts and soil moisture). On the other hand, the magnitude and temporal variability of hydrological processes above this range (cover >50%) are regulated biotically, limiting the impact of abiotic processes, essentially crusting. Although this threshold constitutes an interesting indicator for mining reclamation, we must consider it with care, especially with respect to soil erosion. In fact, the small plot area used for rainfall simulations (0.24 m^2) restricts our observations to splash and sheet flow processes. Nevertheless, cumulative erosion rates recorded in bounded 3x15-m plots (including active rilling) in these artificial slopes from October 2005 to October 2006 also showed a sharp transition between 30-50% cover (Fig. 4.6; data from Moreno-de las Heras *et al*., 2007). Several studies in natural Mediterranean environments have shown that when vegetation cover drops bellow 30% soil erosion and runoff increase drastically (Francis and Thornes, 1990; de Luis *et al*., 2001; Gimeno-García *et al*., 2007). In these natural soils, long-term spatially structured vegetation patterns play an important role in addition to cover, increasing the stability and resilience of the system (Cammeraat and Imeson, 1999; Boer and Puigdefabregas, 2005). The studied man-made system, developed in the context of primary succession, are certainly more vulnerable. In fact, our reported sediment concentrations are relatively high unless 50% ground cover is achieved (Fig. 4.4f) and, at the slope scale, discontinuous rills are still present at 40% vegetation cover (slope 4). Overall results and observations show that a satisfactory biological control of hydrological processes operating in the studied reclaimed soils can be achieved with a minimum vegetation cover of 50%. In fact, there is considerable evidence that the restoration of 50% cover with herbaceous vegetation is decisive for site stabilization in human-made systems of similar characteristics (Andrés and Jorba, 2000; Loch, 2000). This critical threshold can be potentially used as a practical indicator for the evaluation of reclamation works in Mediterranean-Continental environments, where achieving the development of continuous vegetation cover seems to be an extremely difficult goal. The particular analysed conditions (cover of perennial herbs in a 20º sloping Mediterranean mining environment reclaimed with a clay-loam and non-saline substratum) must be considered in order to transfer these results to other areas. Indeed, previous works have stressed a critical influence of site-specific soil characteristics, topography, vegetation type and climate on such vegetative cover thresholds (Snelder and Bryan, 1995; Weltz *et al*., 1998).

Conclusions

The results reported illustrate the strong influence of herbaceous vegetation cover upon the hydrological response of reclaimed mining soils in a Mediterranean-Continental environment (central-eastern Spain). Higher amounts of vegetation cover were associated with a generalised delay in runoff, an increase in soil infiltration capacity and the reduction of soil erodibility. Additional reductions in the influence of seasonal climatic fluctuations on infiltration rates were driven by the biological influence of vegetation cover.

The most dramatic differences were found between areas with under 30% and over 50% vegetation cover, highlighting the transition from the prevalence of abiotic factors to biotic control. We have obtained a practical threshold for ground cover restoration: a minimum of 50% of herbaceous vegetation cover, which enables satisfactory biotic control of soil hydrological responses. This cover threshold could be considered a useful criterion for the evaluation and management of restoration practices in reclaimed Mediterranean-Continental environments.

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Capítulo 5

Capítulo 5

Relaciones de escala de los flujos de escorrentía superficial y de la producción de sedimentos en un gradiente de degradación

Este capítulo reproduce el texto del siguiente manuscrito:

Moreno-de las Heras, M., Wilcox, B.P., Merino-Martín, L., Nicolau, J.M. En preparación. Scale dependency of slope runoff and erosion along a gradient of degradation.

Resumen

Las relaciones de escala y la dinámica espacial de la escorrentía superficial y la erosión del suelo representan cuestiones fundamentales en las áreas de conocimiento de las ciencias geomorfológicas y ecohidrológicas. En ambientes con restricciones hídricas, el estudio de las variaciones que se producen en el espacio en los flujos de escorrentía y sedimentos constituye un punto crítico para el análisis de la organización y el funcionamiento de los ecosistemas de ladera, así como para la comprensión de los procesos de degradación que pueden operar en éstos. Las relaciones de escala de la escorrentía superficial y la erosión del suelo pueden verse afectadas a causa de la activación de procesos degradación de la cubierta vegetal y las características superficiales de las laderas, limitando la redistribución de recursos (agua y sedimentos) en el espacio, proceso del que depende el mantenimiento y desarrollo de la vegetación en ambientes secos. Para comprobar este fenómeno se ha analizado durante un año hidrológico (2005-06) la producción de escorrentía y sedimentos en 20 parcelas de diferente longitud (1-15 m) situadas en cinco laderas mineras de ambiente mediterráneo-seco (Utrillas, España) que representan un gradiente de degradación (de la cubierta vegetal y características de superficie) provocado por procesos acelerados de erosión en regueros. En general, al aumentar la escala (longitud de las parcelas) se han observado reducciones en la producción de escorrentía por unidad de superficie, produciéndose ladera abajo la re-infiltración parcial de los flujos superficiales. No obstante, la eficiencia de los procesos de re-infiltración decrecieron con el nivel de degradación de las laderas. La baja capacidad de los sistemas degradados para ralentizar y almacenar los flujos de escorrentía dentro de las laderas fue especialmente nítida en las precipitaciones de alta intensidad. La relación entre la producción de sedimentos por unidad de superficie y la longitud de las parcelas fue muy variable, cambiando de negativa en el caso de laderas no erosionadas con buen desarrollo vegetal a positiva en las laderas más degradadas, en las que las precipitaciones de alta intensidad incrementaron de forma notable la contribución de la erosión en regueros a la producción total de sedimentos. La pérdida resultante de recursos hídricos y suelo para la vegetación podría explicar el reforzamiento de los procesos de degradación en ecosistemas de ladera intensamente perturbados.

Palabras clave: ambientes secos, erosión en regueros, longitud, re-infiltración, vegetación.

Scale dependency of slope runoff and erosion along a gradient of degradation

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Abstract

Scale relations of slope runoff and erosion represent a major issue for geomorphological and eco-hydrological sciences. In fact, runoff and sediment yield variations througout hillslope systems are recognized as key processes for understanding ecosystem organization and degradation processes in drylands. Degradation of vegetation cover and surface features could modify the scale dependency of runoff and erosion, limiting water and sediment redistribution within the slope. To test this, runoff and sediment yield were monitored on twenty plots of different length (1-15 m), during the 2005-06 hydrological year. These plots were placed on a series of five reclaimed mining slopes affected by a gradient of degradation (cover and surface features) caused by accelerated rill erosion processes in a Mediterranean-dry environment (the Utrillas fieldsite, central-eastern Spain). General decreases of unit-area runoff were reported with scale (plot length), indicating partial re-infiltration of surface flows in the down-slope direction. Nevertheless, the efficiency of runoff re-infiltration along the slope severely decreased with degradation. The low ability to reduce the speed and store overland flow within the slope in degraded conditions was especially patent for high intensity precipitations. The relation of unit-area sediment yield with length was outstandingly variable, changing from negative in well vegetated slopes to positive in severely degraded slopes, where high intensity precipitations increased the contribution of rill erosion to sediment yield. The resulting net loss of soil and water resources for vegetation could be involved in the reinforcement of degradation processes in highly disturbed water-limited hillslope ecosystems.

Key words: drylands, length, re-infiltration, rill erosion, vegetation.

Introduction

Scale issue represents a major challenge in the current state of knowledge of both geomorphological and eco-hydrological sciences (Cammeraat, 2002; Yair and Raz-Yassif, 2004). Indeed, the analysis of runoff and sediment yield variations within scale is recognised as a vital task for slope hydrology (Boardman, 2006; Bracken and Crock, 2007). Moreover, the distribution of water and sediment fluxes throughout hillslope has been stressed as a fundamental process for understanding ecosystem organization (Puidefabregas and Sanchez, 1996; Wilcox and Newman, 2005) and degradation in drylands (Wilcox *et al*., 2003; Cammeraat, 2004; Turnbull *et al*., 2008), especially in the present context of land use and climate changes (Puigdefabregas, 1998; Lavee *et al*., 1998; Sarah, 2004).

It is widely known that runoff per unit-area decreases with slope length because of re-infiltration in the down-slope direction (Yair and Lavee, 1985; Kirkby, 2001; Boix-Fayos *et al*., 2007). Runoff re-infiltration has been primarily linked with the spatial heterogeneity of infiltration and runoff transfer (Yair and Kossovsky, 2002; Cerdan *et al*., 2004). Furthermore, temporal variations of rainfall and surface runoff have a significant role explaining active re-infiltration processes, even in the absence of spatial variability (Wainwright and Parsons, 2002; Kirkby *et al*., 2005; van de Giesen *et al*., 2005).

Soil erosion also depends on scale. Polynomial approaches have been applied to describe and predict general increases of soil erosion with slope length (Wischmeier and Smith, 1978). Nevertheless, there is evidence that no single relationship exists (Morgan, 1995). In fact, several works have pointed reductions in soil erosion with length (Evans, 1995; Wilcox *et al*., 2003; Parsons *et al*., 2006). To explain this disparity, Parsons *et al*. (2004) argument different scale relations depending on the occurrence of sheet and rill erosion.

Runoff re-infiltration and sediment redistribution play a central role in maintaining vegetation structure and organization in a wide range of water-restricted environments (Saco *et al*., 2007). In these systems, the control of resources at the slope scale is efficiently sustained by positive feedback mechanisms between water and vegetation (Puigdefábregas *et al*., 1999; Ludwig *et al*., 2005). Disturbance-induced degradation of vegetation and surface topographical features can have important consequences in hydrological scale relations and hence affect ecosystem function (Turnbull *et al*., 2008). In this way, Wilcox *et al*. (2003) hypothesized that scale relationships of slope runoff and erosion are very different depending on the degree of degradation: in undisturbed hillslope systems unit-area runoff and erosion strongly and non-linearly decrease from small scales to the slope scale, while in disturbed systems these reductions are less pronounced (or even they may turn into increases with scale in the case of sediment yield). The unleashed loss of resources (water and soil) from highly disturbed systems, could reinforce the degradation process, limiting vegetation growth and consequently increasing soil erosion and the loss of water from the hillslope system (Whisenat, 2000).

The objective of this work is documenting the interaction between degradation and scale (length) dependencies of runoff and sediment yield. We state as fundamental assumption the above hypothesis suggested by Wilcox *et al*.

(2003), which advises very different scale relations depending on degradation level. To asses this, we used a multiple scale (1 to 15 m length plots) hydrological data set covering a 12 month period, which was obtained in reclaimed mining slopes affected by a gradient of degradation in Mediterranean-dry Spain.

Materials and Methods

Site Description

This work was carried out in the Utrillas field site, an experimental station located in El Moral, a reclaimed coal-mine in central-eastern Spain (40º47'24'' N, 0º49'48'' W, 1100 m). The climate is Mediterranean-Continental, with a mean annual air temperature of 11ºC (6.8ºC in December and 23.5ºC in July); the air frost period runs from October to April. The local moisture regime can be classified as Mediterranean-dry (*sensu* Papadakis, 1966): Mean potential evapotranspiration is 758 mm and annual precipitation, most of which occurs in spring and autumn, is 466 mm. Especially remarkable in the area is the erosion potential of some events (late spring and summer rainstorms in particular), the intensities of which occasionally reach 100 mm in 24 h (Peña *et al*., 2002).

For this study, five slopes in the reclaimed mining area were monitored. These slopes are all north-facing and have a rectilinear shape with a slope gradient of 20º. They were restored during 1988-89 by the Minas y Ferrocarril de Utrillas S.A. company, through a series of procedures: first, a 100-cm-thick layer of overburden, from the Utrillas cretacic formation of Albian age, was spread over the spoil bank; this overburden has a clay-loam texture (kaolinitic-illitic mineralogy) with a basic pH (Table 5.1) that can be classified as non-saline and non-dispersive according to Rengasamy *et al*. (1984). Next, the soil was prepared for revegetation by cross-slope plowing, to create a transversal pattern of surface roughness that would inhibit rapid overland flow and facilitate water storage in the soil surface. Finally, the slope was sown with a mixture of perennial grasses and leguminous herbs (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis*, *Lolium perenne*, *Medicago sativa* and *Onobrychis viciifolia*; nomenclature following Tutin *et al*., 1964-1980).

In the years following the reclamation of these slopes, degradation has taken place (cover, surface, and soil features), to varying degrees of intensity. Rill erosion has accelerated, mainly because a flawed geomorphological design has enabled excessive amounts of overland flow to be generated from upslope water contributing areas (Table 5.1). Three of the slopes (1, 2 and 3) exhibit dense, well-developed rill networks (0.58-0.95 m m^{-2}). On these slopes the soil surface is bare or sparsely covered (1%-30% total cover) and has lost most of its original roughness. The soils of these highly degraded slopes have a massive and very dense (1.39-1.51 g cm^{-3}) structure and very low organic matter content (0.5%-1.3%), making their infiltration capacity rather low (f_c: 10-20 mm h^{-1}). In contrast, Slope 4 has a discontinuous and poorly developed rill network (0.30 m m^{-2}), and Slope 5 is devoid of rills. These slopes also have more cover, if discontinuous (45%-60% total cover) and retain their original surface roughness. Finally, the soils of slopes 4 and 5 have a lower bulk density (1.23-1.39 g cm^{-3}) and higher infiltration capacity (f_c: 23-37 mm h^{-1}), influenced by the presence of herbaceous roots and higher amounts of organic matter (1.5%-2.0%).

Table 5.1. Basic characteristics (topography, soil, surface traits and erosion features) of the five experimental slopes. Mean ± standard deviation values are showed.

	N	Slope 1	Slope 2	Slope 3	Slope 4	Slope 5
Date of reclamation		1989	1989	1988	1988	1988
Topography						
Slope length (m)		55	50	75	75	60
Slope gradient (°)		20	20	20	20	20
Water-contributing area (m)		8.0	8.0	6.5	4.0	0.0
Aspect		North	North	North	North	North
Soil traits						
Stoniness (%)	25	22.2 ± 2.2	24.7 ± 3.5	26.2 ± 4.1	25.2 ± 2.6	24.5 ± 3.3
Sand (%)	25	33.6 ± 3.6	33.5 ± 3.7	33.8 ± 3.0	39.9 ± 1.8	36.3 ± 2.7
Silt (%)	25	26.9 ± 2.8	33.8 ± 1.6	30.8 ± 1.8	26.4 ± 2.9	26.6 ± 4.5
Clay (%)	25	39.5 ± 2.2	32.8 ± 2.9	35.4 ± 2.1	33.8 ± 2.1	37.1 ± 2.9
Texture	25	Clay loam	Clay loam	Clay loam	Clay loam	Clay loam
pH -H_2O; w/v: ½-	25	8.0 ± 0.1	8.0 ± 0.1	8.0 ± 0.1	8.0 ± 0.1	7.9 ± 0.1
EC -w/v: ½- (dS m^{-1})	25	0.24 ± 0.13	0.26 ± 0.14	0.20 ± 0.10	0.19 ± 0.03	0.23 ± 0.03
CEC ($cmol_c$ kg^{-1})	25	29.5 ± 0.9	23.3 ± 3.3	28.3 ± 1.0	26.0 ± 0.5	22.1 ± 3.5
ESP (%)	25	0.68 ± 0.46	0.27 ± 0.11	0.13 ± 0.01	0.12 ± 0.01	0.18 ± 0.03
Bulk density (g cm^{-3})	75	1.51^a ± 0.14	1.49^a ± 0.12	1.39^a ± 0.17	1.39^a ± 0.12	1.23^b ± 0.17
Organic matter (%)	25	0.58^a ± 0.20	0.56^a ± 0.23	1.27^{ab} ± 0.35	1.46^{ab} ± 0.83	2.00^b ± 0.74
f_c (mm h^{-1}) †	75	11.5^{ab} ± 6.3	10.1^a ± 5.7	20.5^{ab} ± 7.6	22.7^b ± 9.3	36.7^c ± 6.9
Surface traits						
Roughness index††	60	1.04^a ± 0.01	1.05^a ± 0.01	1.10^{ab} ± 0.01	1.15^{bc} ± 0.03	1.23^c ± 0.01
Vegetation cover (%)	150	1.1^a ± 2.0	8.2^{ab} ± 5.5	27.8^b ± 9.9	44.3^{bc} ± 16.2	59.4^c ± 20.8
Erosion features						
Rill density (m m^{-2}) †††		0.95	0.78	0.58	0.30	0.00

Abbreviations: N: number of samples; EC: electrical conductivity; w/v: relation weight (soil) / volume (water); CEC: cation exchange capacity; ESP: exchangeable sodium percentage.
† f_c: final infiltration rate calculated from rainfall simulations (data from Moreno-de las Heras *et al.*, in press).
†† Roughness tortuosity index (following Kamphorst *et al.*, 2000).
††† Linear rill length (m) per surface area (m^2).
Values with the same letters within rows do not differ significantly at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney.

Experimental set-up

From October 2005 to October 2006, runoff and sediment yield were monitored on the five experimental slopes. A number of plots, delimited by 20-cm-high metallic sheets inserted 10 cm into the soil, were established on each slope (Fig. 5.1). In the interrill areas of the slope, three small Gerlach plots were established, each measuring 0.5 m wide: one plot 1 m long (G1), one 2 m long (G2), and one 3 m long (G3). At the downslope end of each plot was a trough, connected to a 50-liter drum for storage of runoff. In addition, a microcatchment plot (MC), 15 m long by 3 m wide, was established on each slope to encompass a representative rill network. At the foot of each microcatchment, two 1.5-m-wide collectors were installed; a cemented central outlet directly connected to the principal rill fed into these collectors. From the collectors, runoff was guided into two 200-liter storage tanks connected by a ten-slot runoff divider.

Runoff was measured and collected from all the plots within 24 hours of each runoff event (during the study, no runoff event exceeded the storage capacity of the tanks and drums). Sediment concentrations were determined

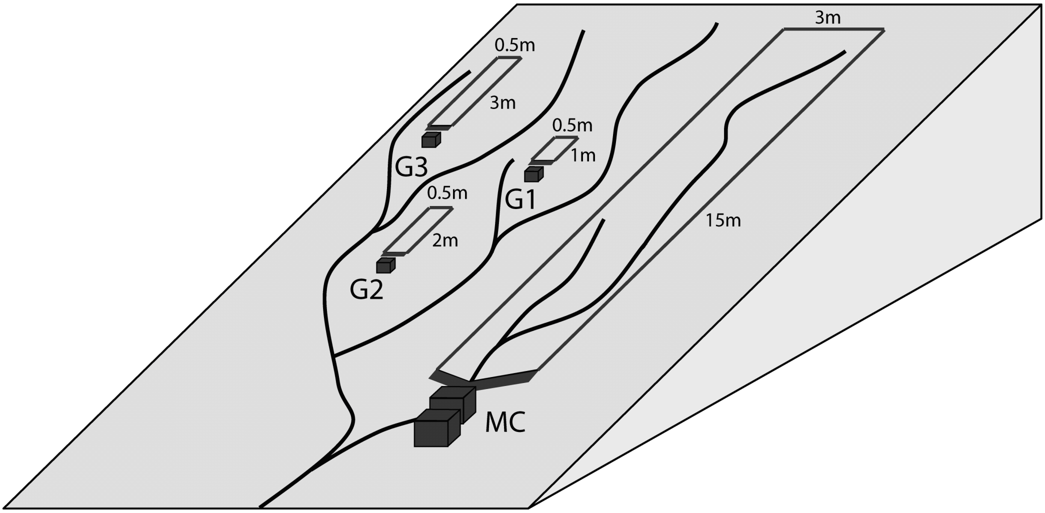

Figure 5.1. Experimental layout. Abbreviations: G1: 1 m length plot; G2: 2 m length plot; G3: 3 m length plot; MC: 15 m length microcatchment.

from 1-liter samples. Precipitation amounts and characteristics were directly measured by an automatic recording rain gauge (GroWeather, Davis®) installed on site.

Vegetation cover and surface roughness were also characterized for each of the runoff plots. Vegetation cover was visually estimated on twenty 0.25-m^2 quadrats distributed over the runoff plots on each experimental slope (one on the G1plot; two on the G2 plot; three on the G3 plot; and 14 on the MC plot). Soil surface roughness was characterized by delineating twelve 1-m-long linear transects in each plot (one for each G1 plot, two for each G2 plot, three for each G3 plot, and six for each MC plot. A 1-m-long pin measure having 50 aligned, mobile needles was used to transfer the soil microtopography in two dimensions onto paper, following the methodology of Bochet *et al*. (2000). The ratio of the surface profile length of each transect to the length of the straight line formed by its projection was then used to calculate a simple surface roughness tortuosity index (Kamphorst *et al*., 2000). The roughness index reaches a minimum value of 1 for ideal smooth surfaces and grows as soil surface roughness increases.

Complementary information was recorded from the areas of each slope outside the runoff plots. Vegetation cover was measured in ten additional 0.25-m^2 quadrats. Soils were analyzed from five composite samples taken from the top 10 cm of the soil profile on each slope (each sample consisted of six subsamples, homogeneously mixed to minimize spatial variability). The analyses followed standardized procedures described in MAPA (1994). Soil bulk density was determined from fifteen unaltered soil cores (3 cm long by 5 cm in diameter) collected from each slope. Finally, soil infiltration capacity was characterized: the final infiltration rate (f_c) was calculated by fitting data from rainfall simulation experiments (15 experiments -each at least 30 minutes long- conducted on 0.24-m^2 plots at 63 mm h^{-1}) to a Horton-type function (Moreno de las Heras *et al*., in press).

Statistical analysis

To identify which precipitation events caused erosion, cluster analysis was used to group rainfall events according to their characteristics (depth, I_1, I_{30} and duration). Data from the larger plots (the microcatchments) of all five experimental slopes were averaged to obtain a mean hydrological response for the study site as a whole. To analyze how plot length influences annual cumulative runoff and sediment yield, the best fitting regression functions were determined. In addition, differences in both runoff and sediment yield between plots of different lengths were analyzed by applying non-parametric Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney tests; we used rainfalls grouped within the different types of precipitations (those previously identified using cluster analysis) as replications (Lal, 1997).

Results

Sources of precipitation

Total rainfall during the 2005-06 hydrologic year was 615.4 mm, about 32% above the annual average. About 30% of rainfall events produced runoff, but most of the runoff and erosion was generated by relatively few events (Table 5.2). Three categories of runoff-producing rainfall events were identified: (1) low-intensity rains occasioned by non-active Atlantic fronts, beginning in the autumn and lasting until the early spring (these rains were the most common); (2) more intense events caused by active Atlantic fronts, occurring during the same period (these rainfall events were an important source of runoff, responsible for about 38% of the annual total); and (3) high-intensity convective thunderstorms, occurring during late spring and summer (these events produced 58% of the runoff and 76% of the erosion).

Runoff

Runoff displayed a great variability, both among and within the five experimental slopes (Table 5.3). The variability among the slopes was related to the extent of degradation: for the most densely rilled slope (Slope 1), both the annual cumulative runoff depth and the runoff coefficient were more than double that of the most vegetated slope (Slope 5).

Table 5.2. Summary of the different sources of precipitation and its average runoff and sediment delivery responses (averaged from 3x15-m microcatchments -MC- in all slopes for each event).

	N	Precipitation (mm)				I_{30} (mm h^{-1})		Runoff (mm)				Sediment yield (g m^{-2})			
		Mean	S.D.	Cum.	%	Mean	S.D.	Mean	S.D.	Cum.	%	Mean	S.D.	Cum.	%
Non-erosive	56	0.5	0.5	26.4	4	0.6	0.5	0.0	0.0	0.0	0	0	0	0	0
Non-active Atlantic fronts	11	9.8	5.3	107.8	18	2.6	1.3	0.4	0.6	3.8	4	4	10	43	1
Active Atlantic fronts	8	34.7	27.2	277.3	45	9.5	2.6	4.2	6.8	33.6	38	144	420	1157	23
Convective thunderstorms	4	50.9	15.5	203.9	33	30.2	16.4	12.8	10.3	51.3	58	947	1227	3786	76
Total	79			615.4	100					88.7	100			4986	100

Abbreviations: N: Number of events; S.D.: Standard deviation; Cum.: Cumulative amount; %: Relative percentage amount.

Table 5.3. Annual cumulative amounts of runoff, sediment yield and global runoff coefficient.

		Cumulated runoff (mm)	Annual runoff coefficient (%)	Cumulated sediment yield (g/m^2)
Slope 1	G1	181.32	29.63	5218
	G2	159.29	25.88	5219
	G3	137.89	22.41	6207
	MC	132.06	21.46	12161
	Mean ± S.D.	152.64 ± 22.42	24.85 ± 3.72	7201 ± 3339
Slope 2	G1	161.03	26.17	5264
	G2	143.23	23.27	5127
	G3	127.60	20.73	6495
	MC	129.28	21.01	8061
	Mean ± S.D.	140.29 ± 15.50	22.80 ± 2.52	6237 ± 1363
Slope 3	G1	143.60	23.34	2824
	G2	116.65	18.96	2654
	G3	101.64	16.52	2762
	MC	97.96	15.92	3270
	Mean ± S.D.	114.96 ± 20.73	18.69 ± 3.37	2653 ± 201
Slope 4	G1	128.08	20.81	2018
	G2	65.42	10.63	1515
	G3	58.76	9.55	1162
	MC	57.34	9.32	1217
	Mean ± S.D.	77.40 ± 33.97	12.58 ± 5.52	1478 ± 392
Slope 5	G1	132.49	21.53	1261
	G2	50.05	8.13	427
	G3	36.67	5.96	279
	MC	27.20	4.42	228
	Mean ± S.D.	61.60 ± 48.18	10.01 ± 7.83	549 ± 482

Abbreviations: G1: 1 m length plot; G2: 2 m length plot; G3: 3 m length plot; MC: 15 m length microcatchment; SD.: Standard deviation.

Within each slope, effects of scale were pronounced. Unit area runoff declined as the plot length increased. (Table 5.3). Nevertheless, the magnitude of this decline varied greatly between the different slope conditions (Fig. 5.2a). Indeed, the decline in runoff was much more sensitive to plot length for the most vegetated slopes (a reduction of 55-79% for slopes 4 and 5) than for the most degraded ones (a reduction of 20-32% for slopes 1, 2, and 3).

The effect of scale on runoff was different for each category of rainfall (Fig. 5.3). In the case of the low-intensity, non-active Atlantic front events, unit-area runoff differed signficantly according to plot length (Fig. 5.3a). In contrast, the effect of plot length on runoff was less pronounced for both the active Atlantic front events and the convective thunderstorms (Fig. 5.5b and 5.5c); for these two categories, the effect was significant only in the case of Slope 5 , the non-rilled and most vegetated slope.

Erosion

Erosion was also highly variable among and within the slopes (Table 5.3). Annual sediment yield from the most degraded slope (Slope 1) was 5 to 50 times higher than from the most vegetated slope (Slope 5), depending on plot length (Table 5.3).

The degree to which plot length affected sedi-

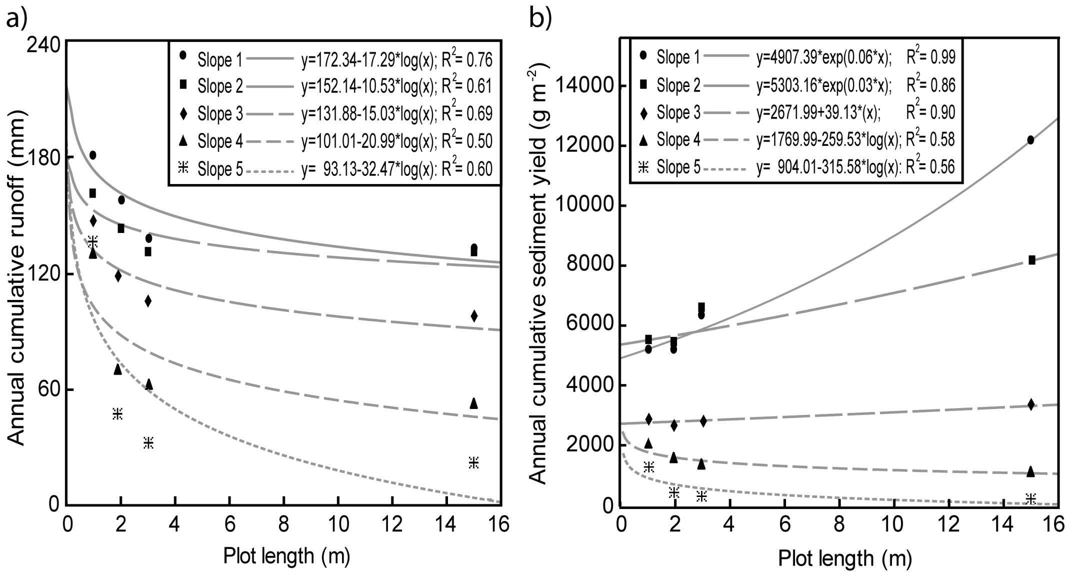

Figure 5.2. Relation between the annual cumulative amounts of (a) runoff and (b) sediment yield and plot length.

ment yield also varied, depending on the extent of degradation (Fig. 5.2b): for the most degraded slopes (Slopes 1, 2 and 3) sediment yield increased with plot length, and these increases were most marked on the longest plots (15-m-long microcatchments), directly influenced by continuous rill networks (rill density: 0.58-0.95 m m^{-2}). In the most severe case (Slope 1), sediment yield from the 15-m-long plot was 133% greater than from the 1-m-long plot. The opposite relationship was found for the less degraded slopes (Slopes 4 and 5), where rill networks were discontinuous or even absent (rill density: 0.00-0.30 m m^{-2}), and sediment was deposited between the rills and in micro-topographical depressions (generated by cross-slope plowing) where vegetation is densely developed: sediment yield from the 15-m-long plots was 39% to 81% lower than that from the 1-m-long plots.

The effect of scale on erosion varied by category of rainfall event, depending on the degree of degradation (Fig. 5.4). For the least degraded slope (Slope 5), sediment yield declined as plot length increased, irrespective of the category of rainfall. Conversely, for the most degraded slope (Slope 1), sediment yield apparently declined with increasing plot length for the low-intensity non-active Atlantic events, but increased significantly with plot length for the high-intensity convective storms.

Discussion

As suggested by Wilcox *et al*. (2003), we found that unit-area runoff is much less sensitive to plot length under degraded conditions: for the most degraded slopes (1, 2, and 3), unit-area runoff from the 15-m-long plots showed a 20-32% decrease compared with the 1-m-long plots, whereas for the less degraded slopes (4 and 5), the corresponding decrease was 55-79%. Similar findings have been previously reported for Mediterranean-dry regions of Spain (a decrease of 26-35% in unit-area

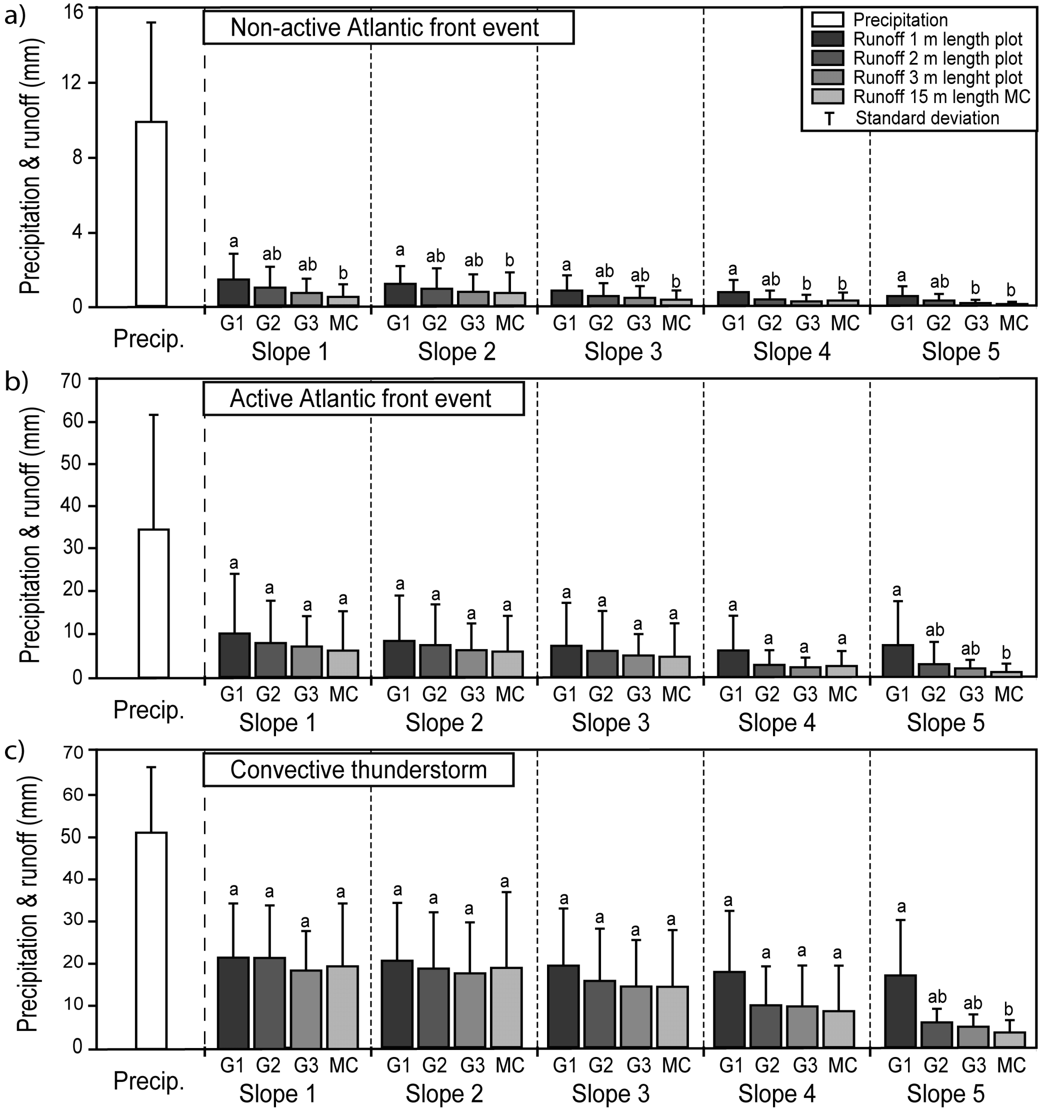

Figure 5.3. Mean runoff generated in the different plots for the three rainfall categories identified [a) non-active Atlantic front type: 11 events; b) active Atlantic front type: 8 events; c) convective thunderstorm type: 4 events]. For comparison, corresponding mean precipitations are showed on the left side. Bars indicate mean values and whiskers indicate standard deviation. Abbreviations: G: 1 m length plot; G2: 2 m length plot; G3: 3 m length plot; MC: 15 m length microcatchment. Bars with the same letters within slopes do not differ significantly at α=0.05. Tested by non-parametric Kruskal-Wallis and *post-hoc* Mann-Whitney.

runoff 30-m-long-plots, compared with small plots of 1.5-3.0 m length, under bare and rilled conditions, *versus* a decrease of 66% under vegetated conditions; Nicolau, 2002); and for subhumid regions of West Africa (a decrease of 40% from 12-m-long-plots, compared with small plots of 1.25 m length, under bare and crusted conditions, *versus* a decrease of 80%

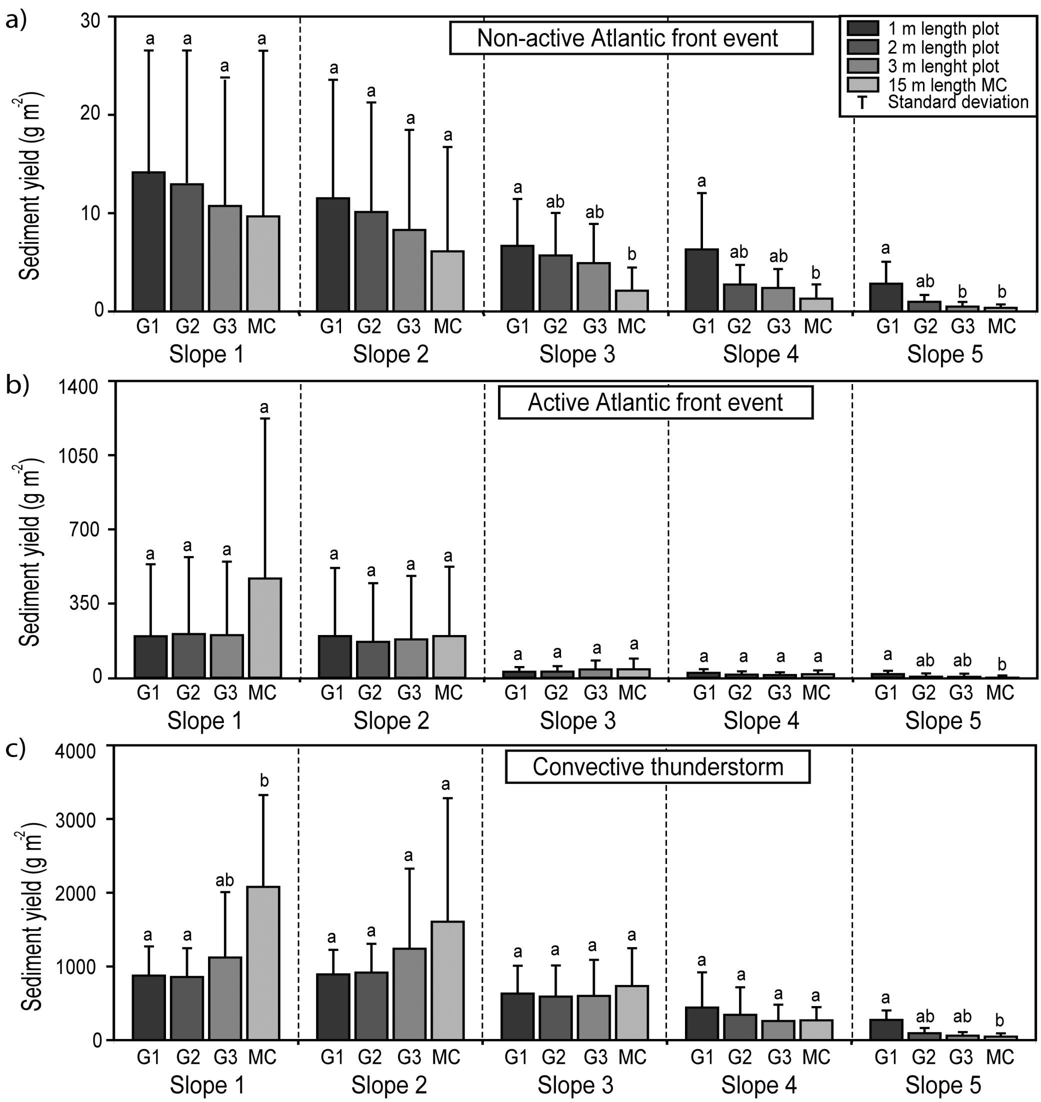

Figure 5.4. Mean sediment yield generated in the different plots for the three rainfall categiries identified [a) non-active Atlantic front type: 11 events; b) active Atlantic front type: 8 events; c) convective thunderstorm type: 4 events]. Bars indicate mean values and whiskers indicate standard deviation. Abbreviations: G: 1 m length plot; G2: 2 m length plot; G3: 3 m length plot; MC: 15 m length microcatchment. Bars with the same letters within slopes do not differ significantly at α=0.05. Tested by non-parametric Kruskal-Wallis and *post-hoc* Mann-Whitney.

under vegetated conditions; van de Giesen *et al*., 2000).

Reductions in unit-area runoff can be attributed to both spatial variability of soil infiltration and temporal runoff dynamics (Wainwright and Parsons, 2002; Yair and Raz-Yassif, 2004; Boix-Fayos *et al*., 2007). The residence time of overland flow on the slope (time required for movement from top to bottom) plays a central role in the efficiency of the process, controlling how long surface runoff has to recharge soil-water stores in the down-slope direction (Tongway and Ludwig, 2001; Joel *et al*., 2002; van de Giesen *et al*., 2005). The extent of degradation of the vegetation strongly influences the capacity of slope systems to slow, retain, and store overland flow. In fact, in the case of the highly degraded slopes in our study (1, 2, and 3), the scarce vegetation cover (less than 30%) meant that surface crusts were well developed, limiting soil infiltration capacity and hence point infiltration. Even more interesting, the residence time of overland flow on slopes like these is much shorter -owing to the inability of the scant vegetation to intercept and slow down the flow- which greatly reduces the efficiency of runoff re-infiltration in the down-slope direction.

Micro-topographical alterations caused by accelerated erosion (the development of rill networks and reduction of surface roughness) also play an important role in runoff re-infiltration processes, by limiting the residence time of flow on the slope. The experimental slopes having well-developed rill networks and rather limited surface roughness (Slopes 1, 2 and 3) showed a low re-infiltration efficiency. A lack of surface roughness offers little resistance to flow, thus severely restricting surface storage (Stomph *et al*., 2002). In addition, the presence of dense rill networks increases runoff connectivity; the rills intercept surface runoff flows throughout the slope and efficiently route them out of the slope system (Nicolau, 2002; Bracken and Crock, 2007).

The low efficiency of runoff re-infiltration processes under degraded conditions was particularly noticeable for the categories of rainfall producing the most runoff. With the low-intensity rainfall events (non-active Atlantic front events), unit-area runoff was scale-dependent for all five experimental slopes. But with the rainfall categories that are the most important sources of runoff (active Atlantic front events and convective storms), scale dependencies were observed only for the most vegetated slope (Slope 5). The degraded slopes showed a remarkable inability to store and delay runoff when storms produced large amounts of overland flow.

The effects of plot length on erosion varied dramatically with level of degradation. For the most degraded slopes (1, 2, and 3), annual sediment yield increased with plot length, in the most severe case showing an increase of 130% from the 15-m-long plot compared with the 1-m-long plot. These increases were due mainly to the erosion caused by concentration of the runoff in rill networks. Conversely, for the most vegetated slopes (4 and 5), sediment yield decreased with plot length. The main process involved in sediment delivery along these vegetated slopes was sheet erosion, as there were no rills (Slope 5) or only poorly developed and discontinuous rills (Slope 4), allowing sediments to be retained within the slope.

Such effects of slope length on erosion rates have generally been attributed to rill erosion processes (Foster *et al*., 1977; Loch, 1996; Parsons *et al*., 2004). In the absence of rill erosion, sediment yield frequently shows no scale dependency, or even a reverse scale dependency brought about the correspondingly lower amounts of runoff per unit area, which limits the entrainment and travel distances of the eroded particles (Parsons *et al*., 1993; Wilcox *et al*., 2003; Parsons *et al*., 2006).

Rainfall category dramatically affected these scale relationships as well. Indeed, the relations observed between annual soil erosion rates and plot length in all the analyzed slopes reproduce in general the up-scaling trends of unit-area sediment yield found for the most intense precipitations, especially the summer thunderstorms -which accounted for 76% of annual erosion-. In the case of the most degraded slopes, the observed up-scaling trend for these high intensity storms -a general increase of soil erosion rates with increasing plot length- disappeared or was reversed for the low intensity non-active Atlantic front events, probably because rilling processes would play a limited role under these rainfall intensity conditions. Liu *et al*. (2000) reported a high dependency of active rilling on rainfall intensity, results that support our finding that the influence of plot length on soil erosion varies with rainfall characteristics.

The overall results of this study are consistent with the conclusions reached by Wilcox *et al*. (2003) regarding the influence of degradation on the scale-dependency of runoff and erosion processes. Indeed, under degraded conditions surface runoff re-infiltration in the down-slope direction was severely constrained, limiting the effect of scale in unit-area runoff. Simultaneously soil erosion rates increased with the scale (length), as consequence of the incorporation of rill erosion processes.

On the whole, degraded conditions led to a

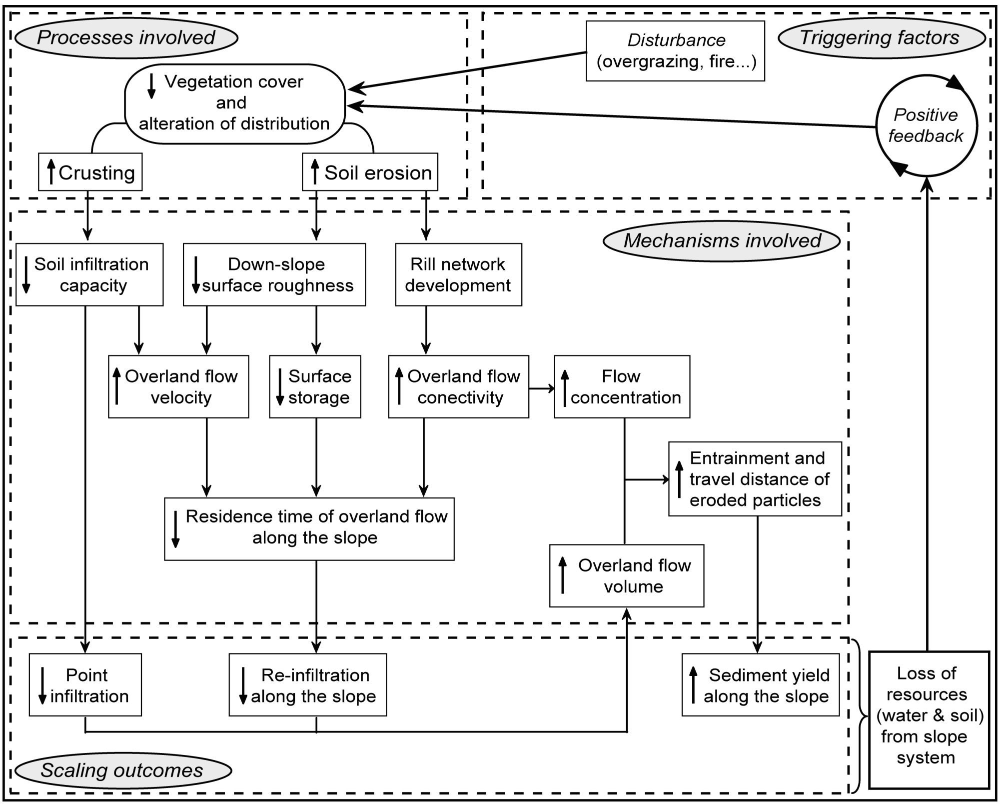

Figure 5.5. Conceptual model summarizing the possible effects of disturbance-induced degradation on hydrological scale dependencies and slope system functionality in water-restricted environments.

substantial loss of resources (water and soil) from the slope system. These results could contribute to our understanding of degradation processes in drylands, as similar mechanisms may operate in disturbed, water-restricted hillslope systems (Fig. 5.5). In fact, there is now ample evidence that the drastic alteration of vegetation cover and distribution occasioned by severe disturbance (e.g., overgrazing, fire, extreme drought) is associated with the loss of soil and water resources from hillslope systems, owing to a decrease in the efficiency of surface runoff re-infiltration and the active contribution of rill and gully erosion (Wilcox *et al*., 1996; Puigdefabregas, 1998). These losses -specifically, the net loss of water resources from hillslope systems- could play an important role in ecosystem functionality, reinforcing the degradation process through feedback mechanisms with vegetation (Davenport *et al*., 1998; Sarah, 2003; Turnbull *et al*., 2008; Espigares *et al*., in press). Indeed, water losses caused by the degradation of vegetation and the acceleration of soil erosion processes could greatly reduce water availability for plant growth, further lowering plant production and cover and thereby further increasing overland flow runoff and erosion (Whisenant, 2002; Wilcox *et al*., 2003).

Conclusions

We have assessed empirical evidence for the hypothesis pointed out by Wilcox *et al*. (2003), who suggested that scale relationships of hillslope runoff and erosion are substantially altered by degradation processes. In fact, the efficiency of runoff re-infiltration along the slope decreases as degradation increases, leading to more continuous and homogeneous flows. Similarly, scale dependence of sediment yield are substantially modified by degradation processes, turning the relations with length from negative to positive; it is mainly caused by the development of rill erosion. The different scaling behaviour of the most degraded slopes is noticeably manifest for high intensity precipitations, due to the remarkable inability to store and delay runoff and the incidence of active rilling processes when large amounts of overland flow occurs.

As a result of overall degradation processes on scaling outcomes, a substantial loss of water and soil resources from the slope goes on. This work could contribute to the understanding of degradation processes in water-restricted environments; in fact, the loss of water resources, as a result of degradation of vegetation cover and distribution and the acceleration of soil erosion processes, from highly disturbed hillslope systems could have important repercussions in ecosystem functionality, affecting vegetation and possibly reinforcing the degradation process.

Acknowledgments

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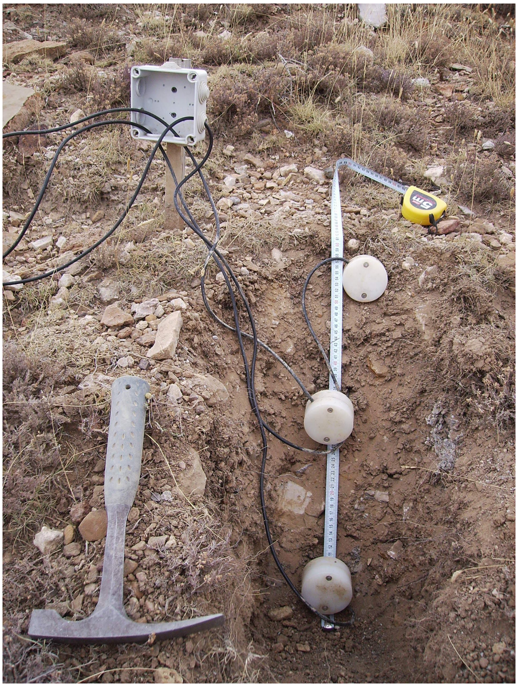
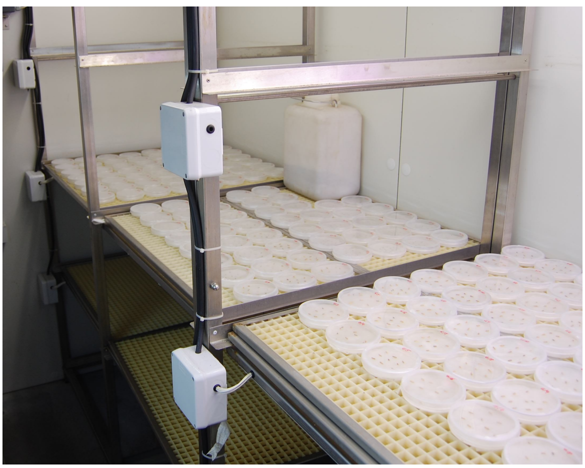
Capítulo 6





Capítulo 6

Efectos hídricos de los procesos de erosión en ecosistemas restaurados de clima mediterráneo-seco

Este capítulo reproduce el texto del siguiente manuscrito:

Moreno-de las Heras, M., Espigares, T., Merino-Martín, L., Nicolau, J.M. En revisión. Water-related ecological impacts of soil erosion processes in Mediterranean-dry reclaimed slopes. Enviado a *Oecologia*.

Resumen

La humedad del suelo está considerada como el principal factor limitante que gobierna la estructura y dinámica de la vegetación en ambientes restaurados de clima mediterráneo-seco. Los procesos de erosión pueden limitar el desarrollo de estos ecosistemas restaurados, especialmente cuando dan lugar a la formación de redes integradas de regueros, ya que éstas pueden condicionar la disponibilidad y distribución espacial de la humedad del suelo. Con el objeto de analizar el impacto de los procesos de erosión sobre la dinámica de los ecosistemas restaurados de clima mediterráneo-seco, se ha estudiado durante el año hidrológico 2005-06 el régimen de humedad del suelo (disponibilidad temporal y espacial) y diferentes respuestas asociadas que describen el estado de la vegetación (potencial hídrico de plantas adultas y germinabilidad de sus semillas en condiciones de campo) y su estructura (riqueza, biomasa y organización espacial) en cinco laderas mineras (cuenca minera de Utrillas, España) sometidas a procesos de erosión en regueros de diferente intensidad (de 0 a 70 t ha^{-1} $año^{-1}$). El desarrollo de las redes de regueros incrementa la conectividad espacial de la escorrentía superficial, concentrando los flujos a lo largo de los canales de drenaje que configuran los regueros. Como resultado, se maximizan las pérdidas de agua de las laderas, y asimismo, la distribución espacial de la humedad del suelo queda gobernada por la distribución de geoformas (regueros e inter-regueros). En respuesta a los condicionantes impuestos por los procesos de erosión sobre la disponibilidad y distribución del agua en el suelo, se produce un aumento de los niveles de estrés hídrico soportados por la vegetación y el desarrollo de condiciones desfavorables para la colonización y el reclutamiento vegetal. Estos efectos se ven traducidos en reducciones de carácter exponencial en la riqueza específica y biomasa de la vegetación asociadas a las tasas de erosión de las laderas. A largo plazo, cuando los procesos de erosión han dado lugar a densas redes de regueros (tasas de erosión en regueros> 20 t ha^{-1} $año^{-1}$), se produce una simplificación drástica de la vegetación, configurando una comunidad poco productiva constituida básicamente por unos pocos individuos de una especie perenne introducida mediante siembra: *Medicago sativa*, la cual presenta una capacidad especial para soportar periodos de sequía intensos. Estos individuos tienden a distribuirse espacialmente en zonas próximas a los regueros, áreas donde minimizan los niveles de estrés hídrico y, simultáneamente, evitan el impacto mecánico generado por los flujos concentrados que discurren por los regueros.

Palabras clave: ambientes secos, disponibilidad de agua, erosión en regueros, estrés hídrico, *Medicago sativa*.

Water-related ecological impacts of soil erosion processes in Mediterranean-dry reclaimed slopes

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Abstract

Soil moisture is considered the main limiting factor governing the structure and dynamics of vegetation in reclaimed Mediterranean-dry environments. Soil erosion is perceived as a critical process affecting these systems, especially when rill formation occurs since rill networks can condition the availability and spatial distribution of soil moisture. To asses the impact of soil erosion processes on the dynamics of Mediterranean-dry reclaimed systems, we monitored during the 2005-06 hydrological year the soil moisture regime (temporal availability and spatial distribution) and the associated responses describing vegetation performance (plant water status and potential seed germination) and structure in five coal-mining reclaimed slopes subjected to different rill erosion rates (from 0 to about 70 t ha^{-1} $year^{-1}$) in central-eastern Spain. Rill network development leads to increase runoff connectivity and concentrate the water flow along the channelling network. As a result, water loss from the slope system is maximized. Simultaneously, the spatial distribution of soil moisture is ruled by the pattern of geomorphic forms (rill and interrill units). The ecological immediate consequences are led by the intensification of drought stress and the occurrence of unfavourable conditions for plant recruitment and natural colonization, causing an exponential decline of species richness and biomass at the slope scale level. When dense rill networks are developed, long-term effects of erosion result in a very simple and low productive plant community basically formed by some drought-resistant *M. sativa* plants (a perennial legume introduced by revegetation). These are spatially organized configuring downward spots adjacent to the rills, where plants minimize at the same time water stress and the mechanical disturbance associated to concentrated flows. A critical degradation threshold might operate in the studied environment at a rill erosion rate close to 20 t ha^{-1} $year^{-1}$, defining the transition to these simplified plant communities.

Key words: drought stress, drylands, *Medicago sativa*, rill erosion, water availability.

Introduction

The analysis of the factors determining the temporal and spatial patterns of soil moisture is a fundamental task for understanding the organization and function of water-limited ecosystems, wherein the interactions between hydrological processes governing soil moisture and vegetation are particularly coupled (Puigdefabregas *et al*., 1999; Wilcox *et al*., 2003). Indeed, water availability is widely recognised as the main controlling resource for vegetation structure and dynamics in both natural and human-made environments under Mediterranean-dry climate (Peco, 1989; Cantón *et al*., 2004; Martinez-Ruiz and Marrs, 2007).

The availability and spatial distribution of this limiting resource can be noticeably altered when accelerated soil erosion occurs, affecting ecosystem dynamics and function (Thornes, 1990; Pimentel *et al*., 1995; Lal, 1998). The loss of water by runoff in highly eroded slope systems is maximized by different mechanisms: the reduction of water infiltration by surface crust formation and cross-slope surface roughness reduction, and the efficient evacuation of water flows from the slope by rill networks, which drastically increase runoff connectivity on the slopes and provide efficient pathways to drive water out of the system (Pimentel and Harvey, 1999; Nicolau, 2002; Bracken and Crok, 2007). Additionally, soil erosion can significantly limit the availability of propagules and soil nutrients (Chambers and McMahon, 1994; Zheng, 2005). Nevertheless, there is evidence that these other effects play a minor role in vegetation dynamics compared to the generally very limited soil water availability (Garcia-Fayos *et al*., 2000; Guerrero-Campo and Montserrat-Martí, 2004; Tormo *et al*., 2006).

Significant reductions of water resources induced by accelerated soil erosion could promote the activation of a long-term self-reinforcing degradation process, reducing plant growth and consequently increasing the intensity of erosion and the associated loss of water by surface runoff (Thornes, 2004; Wilcox *et al*., 2003). In fact, the soil-vegetation system has feedback mechanisms which regulate soil formation, vegetation development and erosion-sedimentation processes (Kirkby, 1998; Puigdefábregas *et al*., 1999). However, although the positive influence of vegetation, increasing infiltration rates and decreasing soil erosion, has been widely documented (Elwell and Stocking, 1976; Francis and Thornes, 1990; Bochet and García-Fayos, 2004) less attention has been paid to the ecological effects of soil erosion (García-Fayos and Cerdá, 1997; Porder *et al*., 2005). In this way, several works indicate a critical role of the knowledge of erosion-vegetation interactions for understanding ecosystem dynamics and degradation processes in water restricted environments, especially in the present context of land use and climate changes (Weltz *et al*., 1998; Wilcox *et al*., 2003).

Reclaimed slopes derived from opencast mining activities are particularly vulnerable to the effects of accelerated soil erosion processes (Loch, 2000; Nicolau and Asensio, 2000). The particular characteristics of freshly reclaimed mining soils (poorly developed, massive structure) in addition to the frequent occurrence of mistakes in the geomorphological design can lead to the genesis and concentration of important amounts of overland flow, promoting soil erosion processes (Ward *et al*., 1983; Nicolau, 2003; Hanckock and Willgoose, 2004). The development of rill and gully networks in these reclaimed systems can drastically limit water availability and modify the spatial distribution of soil moisture at the slope scale, by reducing the opportunities for downslope runoff re-infiltration and concentrating the water flow along the channelling network (Nicolau, 2002; Biemelt *et al*., 2005).

A previous regional work carried out in reclaimed coal-mining slopes of Mediterranean-dry Spain identified rill erosion as a driving force constraining the long-term vegetation succession (Moreno-de las Heras *et al*., 2008). Espigares *et al*. (in press) later documented the impact of soil erosion processes on vegetation performance and dynamics in these reclaimed environments, reducing the diversity and abundance of the soil seed bank and limiting both plant emergence and survival due to increased water stress, suggesting a desiccant filtering effect of rill erosion. Vegetation in these highly eroded reclaimed slopes is represented by a very scarce and simple community, essentially made up of a few aged *Medicago sativa L*. (alfalfa, lucerne) plants growing on interrill areas (avoiding the unstable nature of rill beds) without apparent plant recruitment. This is a perennial legume introduced by revegetation which is able to survive intense periods of water deficit thanks to its extensive tap root system and its ability to remain dormant, shedding leaves and steams, when soil moisture is scarce (Carter and Sheaffer, 1983; Bell *et al*., 2007).

The purpose of this investigation is the analysis of the soil moisture regime (temporal availability and spatial distribution) and ecological implications for reclaimed vegetation in five coal-mining slopes of Mediterranean-dry Spain subjected to different soil erosion intensities since their construction. Therefore, this work intends to improve the understanding of the ecological effects of soil erosion in water-limited environments, applying an ecohydrological perspective. Both species-specific and plant community levels are considered for analysis. In this way, performance and pattern analysis on vegetation (plant water status, potential seed germination in field conditions and plant spatial distribution) specially focuses in the *M. sativa* species, as this is the essential component of vegetation in the analysed eroded landscape, while structural attributes (diversity and biomass) are considered at the slope scale level.

Our main hypothesis is that water availability for vegetation and the spatial distribution of soil moisture are drastically altered by soil erosion processes, especially when rill network development occur. We expect that higher soil erosion rates will affect vegetation by increasing plant water stress and simultaneously reducing potential seed germination in field conditions. Likewise, we expected that plant spatial distribution will reflect the pattern of soil water availability in the slopes.

Materials and methods

Study site

This work was carried out in the Utrillas field site, which is located in the reclaimed mine El Moral (Utrillas coalfield), central-eastern Spain (40º47'24'' N, 0º47'24'' W; 1100 m.a.s.l.). The climate is Mediterranean-Continental. Mean annual air temperature is 11 ºC (6.8 ºC in December and 23.5 ºC in July). The local moisture regime is Mediterranean-dry, according to Papadakis (1966). Mean annual precipitation is 466 mm (concentrated in spring and autumn) and potential evapotranspiration is 758 mm. Vegetation development in this area is constrained by a long frost period (from October to April) and an intense summer drought (from June to October).

The study site consists of five adjacent reclaimed slopes located in the northern side of a spoil-bank. These slopes were restored using very similar treatments during 1988-89 by the Minas y Ferrocarril de Utrillas S.A. company. Slope angle is 20º; the substrate used to cover the spoil-bank is overburden material from the Escucha cretacic formation, of Albian age. This is a clay-loam textured spoil (kaolinitic-illitic mineralogy) with basic pH (Table 6.1). Revegetation was undertaken by sowing a mixture of perennial grasses and leguminous herbs (*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis*, *Lolium perenne*, *Medicago sativa* and *Onobrychis viciifolia*; nomenclature follows Tutin *et al*. 1964-1980).

In spite of having very similar initial features, monitoring of the slopes 2 years after their construction showed differences in the intensity of soil erosion promoted by the accumulation of overland flow in water-contributing areas up-slope, as a result of a wrong design (Nicolau 1996; 2002). Eighteen years of soil erosion processes of different intensity (rill erosion rate from 0 t ha^{-1} year^{-1} in slope 5 up to about 70 t ha^{-1} year^{-1} in slope 1; Table 6.1) have induced large differences in vegetation

Table 6.1. Basic characteristics (Mean ± S.D.) of the five experimental slopes.

	N	Slope 1	Slope 2	Slope 3	Slope 4	Slope 5
Date of reclamation		1989	1989	1988	1988	1988
Topography						
Slope length (m)		55	50	75	75	60
Slope gradient (º)		20	20	20	20	20
Water-contributing area (m)		8.0	8.0	6.5	4.0	0.0
Aspect		North	North	North	North	North
Soil traits						
Stoniness (%)	25	22.2 ± 2.2 a	24.7 ± 3.5 a	26.2 ± 4.1 a	25.2 ± 2.6 a	24.5 ± 3.3 a
Sand (%)	25	33.6 ± 3.6 a	33.5 ± 3.7 a	33.8 ± 3.0 a	39.9 ± 1.8 a	36.3 ± 2.7 a
Silt (%)	25	26.9 ± 2.8 a	33.8 ± 1.6 b	30.8 ± 1.8 ab	26.4 ± 2.9 a	26.6 ± 4.5 ab
Clay (%)	25	39.5 ± 2.2 a	32.8 ± 2.9 b	35.4 ± 2.1 ab	33.8 ± 2.1 ab	37.1 ± 2.9 ab
Texture	25	Clay loam	Clay loam	Clay loam	Clay loam	Clay loam
EC -w/v: ½- (dS m^{-1})	25	0.24 ± 0.13 a	0.26 ± 0.14 a	0.20 ± 0.10 a	0.19 ± 0.03 a	0.23 ± 0.03 a
pH -H_2O; w/v: ½-	25	8.03 ± 0.12 a	7.96 ± 0.14 a	7.95 ± 0.13 a	7.95 ± 0.13 a	7.91 ± 0.10 a
Organic matter (%)	25	0.58 ± 0.20 b	0.56 ± 0.23 b	1.27 ± 0.35 ab	1.46 ± 0.83 ab	2.00 ± 0.74 a
Bulk density (g cm^{-3})	75	1.51 ± 0.14 a	1.49 ± 0.12 a	1.39 ± 0.17 a	1.39 ± 0.12 a	1.23 ± 0.17 b
Vegetation traits						
Vegetation cover (%)	150	1.1 ± 2.0 c	8.2 ± 5.5 c	27.8 ± 9.9 b	44.3 ± 16.2 ab	59.4 ± 20.8 a
M. sativa rel. abund. (%)	5	83.1	75.3	39.7	21.3	11.1
Hydrological features						
Rill density (m m^{-2})	15	0.95	0.78	0.58	0.30	0.0
Rill erosion rate (t ha^{-1}year^{-1})	5	71.41	45.03	16.95	7.86	0.00
Interrill fc (mm h^{-1})*	75	11.5 ± 6.3 ab	10.1 ± 5.7 a	20.5 ± 7.6 ab	22.7 ± 9.3 bc	36.7 ± 6.9 c
2005-06 $Qc_{15m\ plot}$ (%)*	5	21.46	21.01	15.92	9.32	4.42
2005-06 $Rri_{1\text{-}15m\ plots}$ (%)*	5	27.17	19.72	31.78	55.23	79.47

Abbreviations: N: number of samples; EC: electrical conductivity; w/v: relation weight (soil) / volume (water); rel abund.: relative abundance; *fc*: hortonian final infiltration rate; *Qc*: runoff rate; *Rri*: runoff re-infiltration rate.
*Data from Moreno-de las Heras et al., 2007.
Values with the same letters (a-c) within rows do not differ significantly at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney.

development (total vegetation cover ranging from about 1% in slope 1 up to 60% in slope 5), vegetation composition (relative abundance of *M. sativa* ranging from about 80% in slope 1 to 10% in slope 5) and associated soil traits (soil organic matter and bulk density) in these five slopes, following a degradation gradient (Table 6.1). Although these five slopes do not constitute true replicates, as it is common in mining-reclamation studies, they constitute an unique scenario to analyze the long-term interactions between soil erosion and the dynamics of vegetation (Espigares *et al*., in press).

Characterization of soil, vegetation and hydrology of the experimental slopes

In order to characterize soil traits, five composite soil samples were collected from the top 10 cm in each experimental slope (each formed by six subsamples distributed at random, which were homogeneously mixed in attempt to minimise spatial variability). Soil samples were air dried and sieved (2 mm sieve). General physico-chemical characteristics were determined using standardised methods proposed by the Spanish Ministry of Agriculture (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 the permanent willing point (Ψ= 0; -0.01, -0.03; -1.50 MPa). Parameterisation of soil water retention characteristic curves was made according to van Genuchten (1980). Additionally, soil bulk density (g cm^{-3}) was determined at the top of the soil profile from fifteen unaltered soil sample cores (3 cm height by 5 cm diameter) collected at random in each slope.

Vegetation determinations took place in spring 2006. In each slope, vegetation cover (%) and composition was determined in thirteen 0.50 x 0.50-m plots, distributed at random. *M. sativa* relative abundance (%) was calculated as the ratio between alfalfa cover and total vegetation cover. Species richness was assessed by the total of species. Aboveground biomass was collected in six additional 0.50 x 0.50-m plots and subsequently oven-dried (60ºC; 72 h) and weighted (g m^{-2}).

To asses soil erosion processes, rill density was measured in each slope (m of linear rill/ m^2 of area). Rill sections (width and depth) were determined in all rills intercepted by three equidistant cross-slope transects of 30 m length. Historical averaged rill erosion rate (t ha^{-1} $year^{-1}$) in each slope was quantified using the former rill network dimensions as well as the bulk soil density and slope age from reclamation (Morgan, 1995).

Slope hydrology is illustrated by a series of parameters (interrill infiltration capacity, annual cumulated runoff rate and runoff re-infiltration rate) obtained from October 2005 to October 2006 (Table 6.1; data from Moreno-de las Heras *et al*., 2007). The interrill infiltration capacity is described by the steady infiltration rate (Interrill *fc*, mm h^{-1}), obtained from rainfall simulation data experiments (15 experiments in each slope on 0.24-m^2 plots at 63 mm h^{-1} rainfall intensity). Cumulative runoff rates, measured as percentage of total precipitation (2005-06 *Qc*, %), were recorded in bounded 3 x 15-m (wide x length) runoff plots containing representative rill network sections of the slopes. Finally, runoff re-infiltration rate along the slopes (2005-06 *Rri*, %) was assessed by the scale variation of unit-area runoff comprised between cumulated runoff amounts obtained in 0.5 x 1-m and 3 x 15-m bounded runoff plots.

Soil moisture dynamics

During the 2005-06 hydrological year (from October 2005 to October 2006), soil moisture (%, v/v) was monitored using Time Domain Reflectrometry technology (hereafter TDR). A network of sensors, placed on different positions (interrill and rill geomorphic units) and soil depths within the slopes, was used. In each slope, TDR sensors were installed in eight soil profiles: four placed on interrill units and another four on rill microsites (except in slope 5 where rill networks were absent). On interrill soil profiles, TDR sensors were horizontally inserted at 5; 25; 50 and 80 cm depth; on rill profiles, sensors were inserted at 5; 25 and 50 cm depth. Soil moisture determinations were done periodically (each 15 days without rain and 24h after each rainfall event) using a TDR cable tester (Tektronix® 1502C), following the guidelines of Cassel *et al*. (1994).

Plant water status

To analyze the water status of vegetation growing in the studied slopes, two campaigns of water potential measurements were carried out in mid June and late August 2006, when soil moisture reached the lowest values. Leaf water potential (Ψ_l, MPa) of *M. sativa* plants was determined using a pressure chamber (SKPM 1400, Skye Instruments®), following the methodology proposed by Brown and Tanner (1981). Water potential measurements were restricted to three of the five studied slopes (slopes 2; 3 and 5), as the long operation time required made impossible to obtain a representative minimum of samples in all slopes. A threefactorial approach was adopted, taking into account slope, position and time. Thus, to asses the influence of rill networks in the water status of vegetation (by spatially distributing soil moisture), position within the interrills was considered: Central and Lateral position (distant and closed to rills respectivelly), as no *M. sativa* plant was found inside the rills. Additionally, two different times were considered: pre-dawn (from 05:00 to 07:00 h) and mid-day (from 14:00 to 16:00 h). In global, 24 measurements homogeneously distributed by position and time were done per campaign in slopes 2 and 3, while only 12 measurements per campaign were done in slope 5, as position was not considered (rills are absent in this slope).

Plant spatial distribution

To asses the impact of rill networks on the spatial organization of vegetation, the distribution of *M. sativa* plants was analysed in the rilled slopes (slopes 1, 2, 3 and 4). In each slope, 200 quadrats (25 x 25-cm) were randomly distributed within the interrills (no *M. sativa* plant was found inside the rills). The number of plants which appeared inside each quadrat (plant abundance) was counted. Additionally, the distance to the nearest rill was measured, as well as the width of the interrill where each quadrat was placed. A new variable which describes quadrat position within interrill areas (plant centrality on interrill) was calculated by dividing the distance to the nearest rill by the half of interrill width (which represents the maximum possible distance to the nearest rill). Thus, plant centrality adopt values ranging from 1 (quadrats placed just in the center of the interrill) to about 0 (quadrats in lateral position, really close to a rill).

Seed germination under different water potentials

Medicago sativa seeds of the variety used during the restoration of the Utrillas fieldsite, ("Tierra de Campos"; Fombellida 2001), were obtained from a local seed supplier (Semillas

Batlle S.A.), due to the impossibility to collect them in the field, as seed production is very low. Germination of alfalfa seeds was studied under eight different water potentials (Ψ= 0.00; -0.03; -0.10; -0.20; -0.33; -0.62; -1.10 and -1.50 MPa) representing soil moisture values between saturation and the permanent willing point. Water potentials were simulated using polyethylene glycol concentrations (PEG-6000) following the standard equations of Michel *et al*. (1983). Ten replicates of each water potential were prepared. Each replicate consisted of 25 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 37 ml of distilled water (control; Ψ= 0.00 MPa) or PEG solutions (Ψ ranging from -0.03 to -1.50 MPa). The experiment took place inside a phytotron under controlled conditions (day length= 12 h; air temperature= 20ºC; relative air humidity= 75%). The experiment lasted 34 days. To avoid water potential variations Petri dishes were sealed with PVC sheets (reducing the loss of water) and, after the first 15 days, replicates were transferred to fresh solutions. Seed germination was monitored every three days; a seed was considered germinated when at least 2 mm of radicle had emerged. Germination rate (%) was calculated as the percentage of germinated seeds during the experiment.

Data analysis

In order to identify the main differences in the basic characteristics of the analysed slopes, Kruskall-Wallis ANOVA and *post-hoc* Man-Whitney U tests were used.

Differences between and within slopes (rill and interill geomorphic units) on 2005-06 soil moisture dynamics has been analysed using repeated measures ANOVA, with time as within subjects factor and slope and geomorphic units as between subjects factors. A special attention has been paid on interrill soil moisture dynamics, as these were the areas where vegetation was actually present (it must be stressed that rill microsites have hardly any vegetation cover due to physical disturbance provoked by concentrated flows). Thus, to obtain a synthetic indicator of the impact of soil erosion processes on the soil water content available for vegetation at the slope scale, the best fitting regression function was determined between the historical rates of rill erosion acting in the slopes and the averaged soil moisture measured on interrill soil profiles after rainfall occurrence (24h after each rainfall event) along the monitored year.

Differences on alfalfa leaf water potential were independently analysed for the two sampling campaigns (mid June and late August). Factorial ANOVA and *post-hoc* Tukey tests were used. Slope, sampling time (pre-down and mid-day) and plant position within interrill areas (central and lateral position) were considered subject factors for the analysis.

The influence of rill networks on the spatial distribution of alfalfa plants was analysed using Spearman's R correlation coefficient between plant abundance and plant centrality, which describes relative plant position within interrills.

To analyse the effect of soil water potential on the germination rate of alfalfa seeds, a classical sigmoid shape response function (Ahmadi and Ardekani, 2006) was fitted. From this relation we determined the water potential threshold values for seed germination. These threshold values were transformed into soil water content using the parameterised characteristic curves of soil water retention. We used the obtained soil moisture threshold values to analyse the potential germination performance on field conditions in the five experimental slopes. Additionally, we considered a temperature threshold value for seed germination of 10ºC air temperatue, since *M. sativa* seeds (particularly those of the studied variety) show a sharp decrease of germinability below this temperature (Brar *et al*., 1991; Fombellida, 2001). Thus, to analyse the potential performance on field conditions we used both soil moisture values on interril areas (where vegetation is actually growing) at 5 cm depth and daily mean air temperature measured during the 2005-06 hydrological year (2005-06, Pp= 615mm). Additional data (only available for slope 2) from a dryer hydrological year (1989-90, Pp= 270mm; Nicolau, 2002) was used to analyse the potential performance in a very contrasted situation.

Finally, the impact of soil erosion processes on vegetation at the slope scale level was assessed by determining the best fitting regression equation between the historical rates of rill erosion and two structural community attributes (species richness and aboveground biomass).

All statistics have been carried out using the STATISTICA 6.0 package (Statsoft 2001).

Results

Soil moisture

Soil moisture dynamics during the 2005-06 year followed the annual pattern of precipitations and temperature (Fig. 6.1). Soil moisture recharge was concentrated mainly during autumn and winter months; nevertheless, some particular increases of soil moisture were generated by summer storms. Major evapotranspiration losses took place in spring and summer.

The spatial distribution of soil moisture was associated to the pattern of rills and interrills (Fig. 6.1). In general, higher soil water contents where observed on rill than on interrill units ($F_{1, 72}$= 46.43, p<0.01). However, soil moisture was differently affected by the type of geomorphic unit in each of the four rilled slopes (slope x geomorphic unit: $F_{3, 72}$= 3.37, p<0.05). In fact, soil moisture was significantly higher on rill soil profiles in the most densely rilled slopes (slope 1: $F_{1, 18}$= 27.06, p<0.01; slope 2: $F_{1, 18}$= 36.91, p<0.01), while these differences were less drastic or even disappeared in the two slopes with discontinuous rill networks (slope 3: $F_{1, 18}$= 10.51, p<0.05; slope 4: $F_{1, 18}$= 1.36, p=0.26).

No differences in soil moisture were found between the slopes on rill areas ($F_{3, 44}$= 0.55, p= 0.65; Fig. 6.1). On the other hand, important differences in soil moisture were found between the five experimental slopes on interrill areas ($F_{4, 60}$= 6.15, p<0.01). In this way, soil moisture inputs on interrill soil profiles were much lower in the most eroded slopes (slopes 1, 2 and 3), while in slopes 4 and 5 these inputs where higher and deeper (Fig. 6.1). This effect is well represented by a non-linear shape function with the soil erosion rate of the studied slopes (Fig. 6.2), indicating a reduction of soil water inputs on interrill areas of about 25% in the most degraded conditions (slopes 1 and 2: 40-70 t ha^{-1} $year^{-1}$ rill erosion) compared to the uneroded slope (slope 5: 0 t ha^{-1} $year^{-1}$).

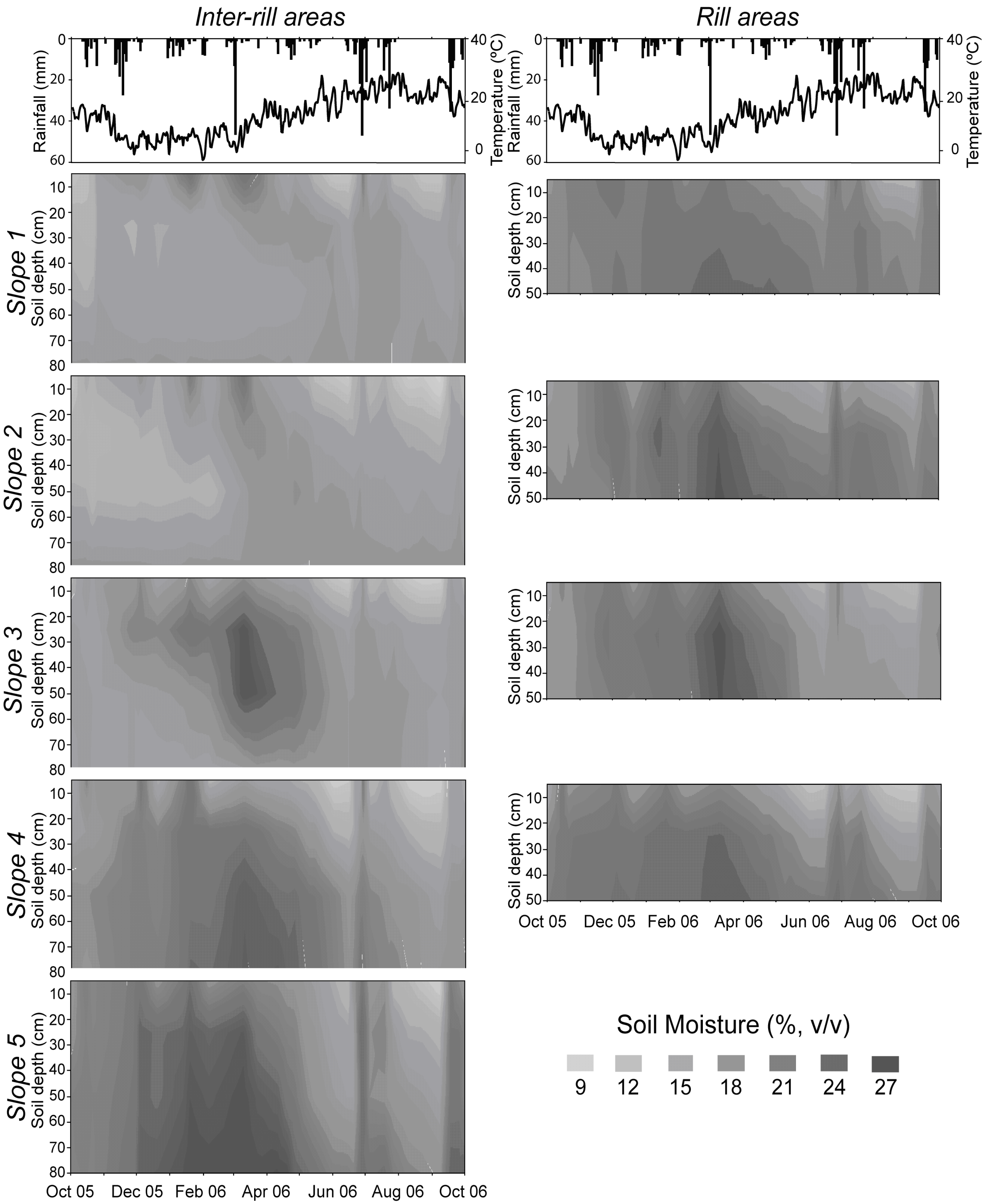

Figure 6.1. 2005-06 soil moisture dynamics in the five experimental slopes. Grey-scaled graphs represent mean soil moisture dynamics of the interrill (left-side) and rill (right-side) soil profiles along the hydrological year. Meteorological data (daily precipitation and daily mean air temperature) along the monitored period is represented by the top-side graphs.

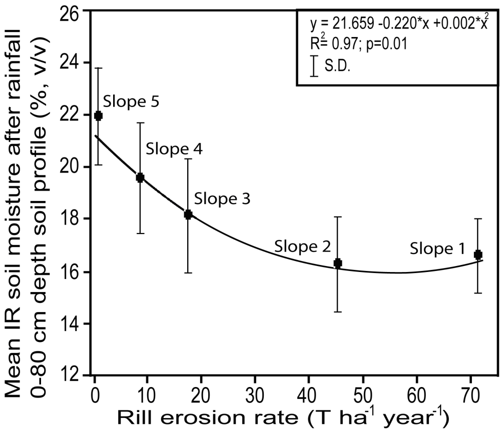

Figure 6.2. Diagram representing the relation between soil moisture inputs on interrill soil profiles (annual mean soil moisture measured 24h after rainfall occurrence) and the soil erosion rates of the slopes.

Plant water status during the seasonal drought period

All *M. sativa* leaf water potential determinations were below -1.5 MPa (Fig. 6.3), indicating water stressed plant conditions during the two sampling campaigns (late spring in June and summer in August). Differences in leaf water potential between the analysed slopes showed the same pattern in both campaigns and measurement pulses (pre-dawn and mid-day): plants in the uneroded slope (slope 5) reached higher leaf water potential values than in the eroded slopes (slopes 2 and 3), where plants were significantly more water-stressed (June: $F_{2,\ 54}$= 11.15, p<0.01; August: $F_{2,\ 54}$=23.45, p<0.01; Fig. 6.3a and 6.3c). Worthy of notice are summer leaf water potentials reached in the eroded slopes (between -4.0 and -5.5 MPa), illustrating the intense drought stress borne by vegetation in these conditions.

Plant position within interrill areas also showed a significant effect in the rilled slopes analysed (slopes 2 and 3): plants growing in central position were more water-stressed than those growing in lateral positions (close to a rill), which generally showed higher pressure values (June: $F_{1,\ 40}$= 34.79, p<0.01; August: $F_{1,\ 40}$= 26.28; p<0.01; Fig. 6.3b and 6.3d). These differences were specially remarkable for mid-day leaf water potential.

Plant spatial distribution

In the most densely rilled slopes (slopes 1 and 2), the abundance of *M. sativa* plants showed a strong negative correlation with plant centrality on interrill areas (Table 6.2). Thus, in these slopes the abundance of growing alfalfa plants is higher on interrill borders, close to the rills, and decrease toward the middle of interrill areas. On the other hand, no significant correlation between plant abundance and relative plant position was found in slopes 3 and 4, where rill networks are discontinuously structured.

Potential plant germination performance on field conditions

Data from the germination experiment under controlled conditions evidenced a drastic effect of water potential on germination rate of *M. sativa* seeds (Fig. 6.4). Germination rate decreased sharply at water potential values ranging from -0.20 MPa to -0.60 MPa (Fig. 6.4a). Over this range seed germination was almost complete (*circa* 90%) while below this was almost negligible (*c.* 0%). These absolute rates of germination were obtained in a period of time ranging from 15 to 25 days from the

Table 6.2. Spearman's R correlation coefficients between *M. sativa* plant abundance and plant centrality on interrill for the studied rilled slopes (slope 5 is voided of rills).

	Spearman’s R	*p*
Slope 1	-0.331	0.000
Slope 2	-0.324	0.000
Slope 3	-0.073	0.272
Slope 4	-0.100	0.145

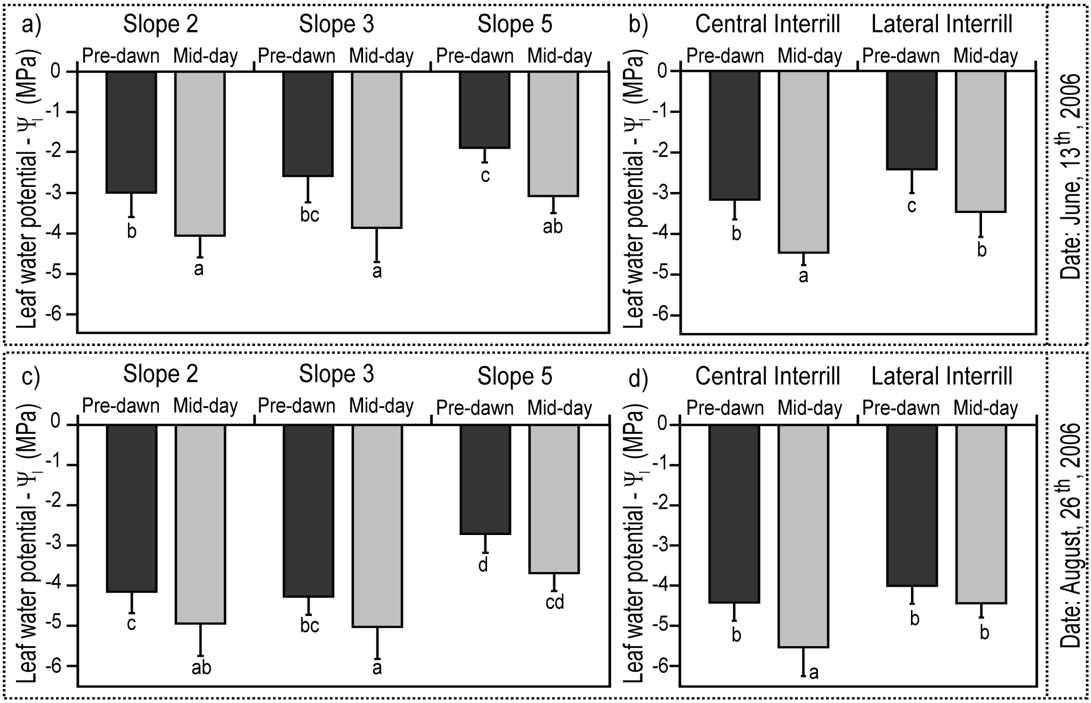

Figure 6.3. *M. sativa* leaf water potential values (bars represent mean values and whiskers the standard deviation) obtained during the two measurement campaigns (mid June and late August). Left-side graphs (a and c) represent differences between slopes; right-side graphs (b and d) represent differences related to interrill plant position on the rilled slopes (slopes 2 and 3). Different letters (a-c) indicate differences at α=0.05. Tested using factorial ANOVA and *post-hoc* Tukey tests.

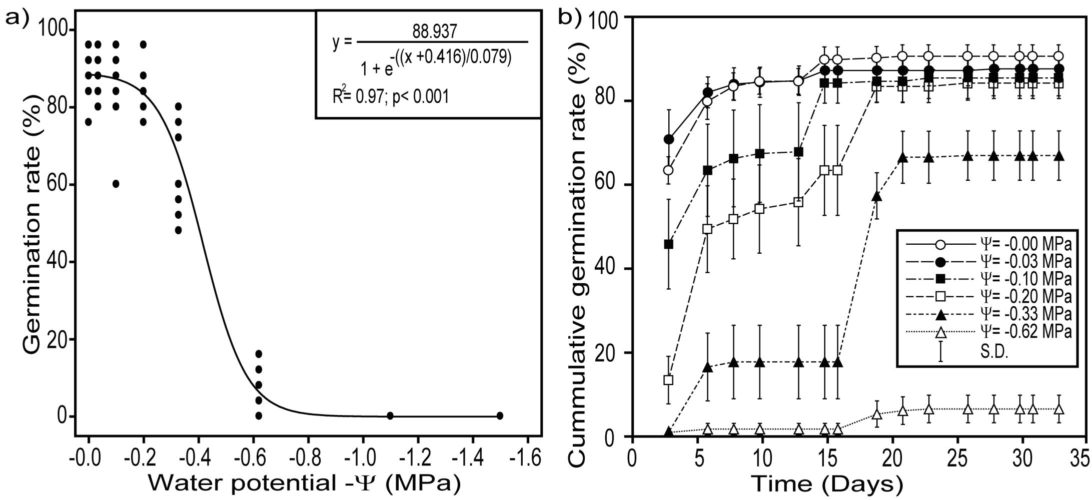

Figure 6.4. Diagrams representing the relation between (a) *M. sativa* seed germination rate and water potential, and (b) *M. sativa* cumulative seed germination rate and experimental time (days).

experiment start (Fig. 6.4b). Water potential values defining the obtained experimental threshold for seed germination (-0.20 MPa and -0.60 MPa) correspond respectively to 16.76% (S.D. 1.56%) and 13.23% (S.D. 1.38%) volumetric soil moisture values. These values are reasonably homogeneous between the analysed soil samples of the five experimental slopes, as no differences of soil moisture between slopes were detected at these pressure values (Ψ= -0.20 MPa: $F_{4, 20}$= 0.90, p= 0.48; Ψ= -0.60 MPa: $F_{4, 20}$= 0.78, p= 0.55).

The 2005-06 hydrologic year was fairly humid as total precipitation was 615 mm (about 32% above the annual average). Daily mean air temperature remained bellow 10ºC during late autumn and winter (from early November to late March; Fig. 6.5a). Thus, throughout this period potential seed germination in the five

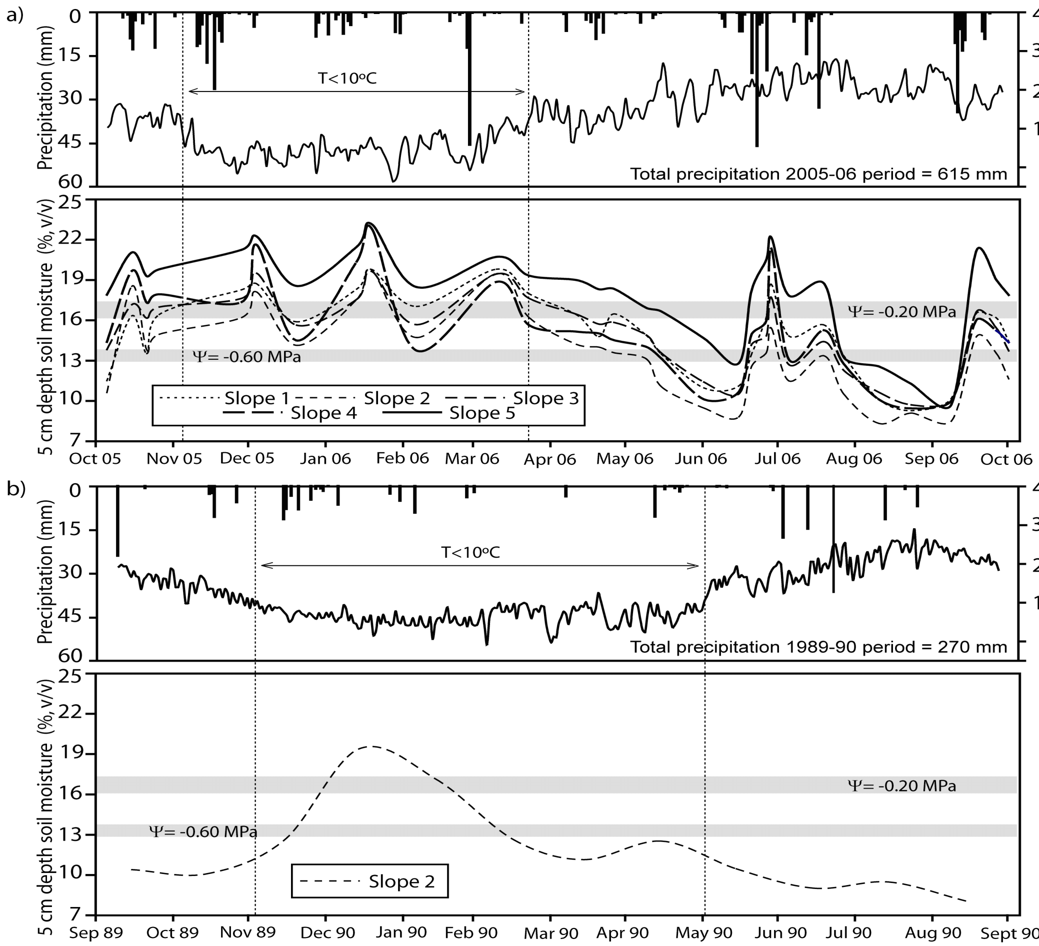

Figure 6.5. Diagrams representing field conditions (air temperature and soil moisture at 5 cm depth) for *M. sativa* seed germination along (a) 2005-06 (all slopes included) and (b) 1989-90 (only slope 2 available; data from Nicolau, 2002). Critical air temperature threshold for seed germination (bellow 10ºC no germination is expected *sensu* Brar *et al.*, 1991 and Fombellida, 1991) is delimited by vertical dotted lines. The obtained critical water potential threshold values for seed germination (over -0.20MPa seed germination is successful, while below -0.60MPa is negligible) are represented by horizontal grey bars (mean ± SD).

experimental slopes was severely constrained because of temperature limitations (Brar *et al*., 1991; Fombellida, 2001). Before this period, soil moisture at 5 cm depth reached values higher or equal soil water potential levels of -0.20 MPa for more than 20 days in slopes 4 and 5. In the most eroded slopes (slopes 1, 2 and 3) these moisture values were reached only occasionally; nevertheless the -0.60 MPa water potential threshold was surpassed for a long period (Fig. 6.5a). Moisture dynamics, after air temperature exceeded 10ºC in spring, followed a similar pattern than in early autumn (Fig. 6.5a): soil moisture in the uneroded slope (slope 5) surpassed the -0.20 MPa water potential threshold during a long time (more than 40 days), while soil moisture in the eroded slopes (slopes 1, 2, 3 and 4) remained between -0.20 and -0.60 MPa water potential range values during about two months. Thus, field conditions for *M. sativa* seed germination during this humid year were very favourable in the uneroded slope (slope 5). In highly and medium eroded slopes (slopes 1, 2, 3 and 4) field conditions were less favourable and probably allowed a limited seed germination and seedling emergence.

In contrast with the described year, the 1989-90 hydrologic year was markedly drier, with a total precipitation of 270 mm (a 42% bellow annual average). Soil moisture in the monitored slope (slope 2) only exceeded the -0.60 MPa water potential threshold during winter (from mid November to mid February) when air temperature remained below 10ºC (Fig. 6.5b). Thus, potential *M. sativa* seed germination was drastically constrained in this highly eroded slope along this dry year.

Relation between soil erosion and slope scale plant community attributes

The analysed community attributes were strongly related to the intensity of the rill erosion processes acting in the experimental slopes since their construction, indicating drastic non-linear decreases of species richness and aboveground biomass with increasing soil erosion (Fig. 6.6). In fact, contrasting with the productive and diverse vegetation composition developed in the uneroded slope (slope 5:

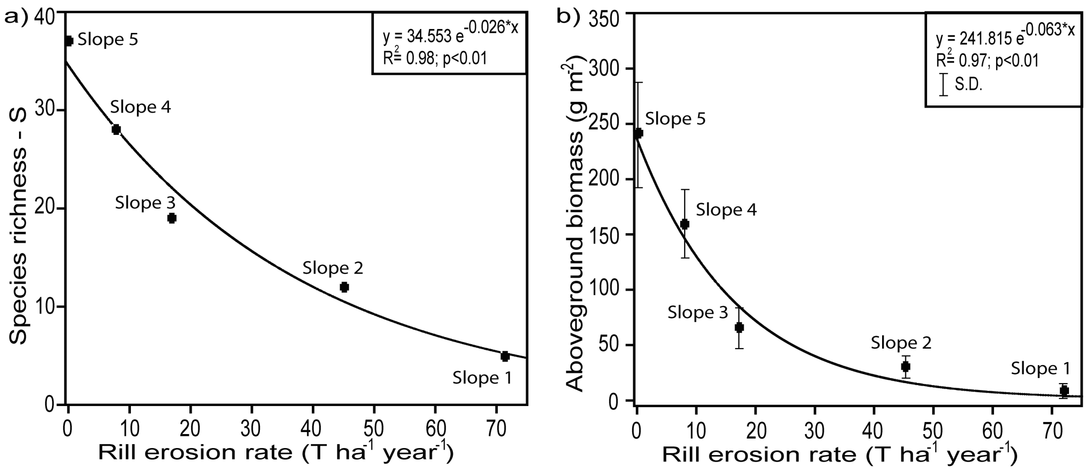

Figure 6.6. Diagrams representing the relations between the soil erosion rates of the slopes and slope scale (a) species richness and (b) aboveground biomass (g m^{-2}).

Biomass= 240 g m^{-2}; S= 37 species), both ecological attributes showed very low values in the most eroded slopes (slopes 1 and 2: Biomass= 10-30 g m^{-2}; S= 5-12 species).

Discussion

Soil moisture dynamics in the studied reclaimed coal-mining slopes reveals an important influence of soil erosion processes in both water availability and spatial distribution of soil moisture, with important consequences for long-term vegetation performance and structure.

The emergence of a general geomorphic pattern constituted by rill and interrill units that limits water availability for vegetation is an elemental process structuring water resources at the slope scale in the eroded slopes studied. This effect is especially evident in the case of the most eroded slopes (slopes 1 and 2; rill erosion rate >45 t ha^{-1} $year^{-1}$). There, soil moisture penetration on interrill areas (where vegetation development is not hampered by the mechanical disturbance caused by concentrated water flows) is severely limited, while water resources are concentrated along the rill networks. This distribution is expected in highly eroded landscapes, where moisture penetration is usually shallow and generally pattered by the presence of runoff routing channelling forms which, at the same time, drive important amounts of water resources out of the slope system (van den Elsen *et al*., 2003; Cantón *et al*., 2004; Biemelt *et al*., 2005). In accordance, other works carried out in this experimental site (Moreno-de las Heras *et al*., 2007) indicated that the pattern of generation and circulation of water flows in these highly eroded slopes (slopes 1 and 2) leads to an acute loss of water resources by surface runoff (more than 20% of precipitation during 2005-06 hydrological year, Table 6.1). These losses are ruled by the lack of vegetation influence on soil conditions, which limits infiltration capacity on interrills (*fc circa* 10 mm h^{-1}; Table 6.1), as well as the high runoff connectivity imposed by the dense rill networks, which dramatically constrains runoff re-infiltration processes along the slope (2005-06$Rri_{1\text{-}15m\ plots}$< 30%, Table 6.1). This situation drastically contrasts with the uneroded slope (slope 5), where vegetation is discontinuously developed (*c*. 60% cover) and runoff (as sheet flow) is widely redistributed along the slope (2005-06$Rri_{1\text{-}15m\ plots}$ *c*. 80%, Table 6.1), limiting the losses of water resources by surface runoff at about 5% of precipitation.

The immediate ecological consequences of the soil erosion processes acting in the studied Mediterranean-dry environment are led by the increase of drought stress related to the associated loss of water resources. We observed higher levels of plant water stress in the eroded slopes along the drought period (late spring and summer), as well as a non-linear decline in aboveground biomass parallel to the erosion-related water availability reduction. These outcomes are comparable with the general results obtained in cropping systems affected by accelerated erosion processes, which point at an accentuation of drought stress and productivity decline associated to the loss of water by surface runoff (Lal, 1998; Pimentel and Harvey, 1999).

It is not by chance that vegetation composition in the most eroded analyzed slopes (slopes 1 and 2) is almost restricted to *M. sativa* species (more than 75% of total cover; Table 6.1), as this perennial legume has demonstrated a special ability to resist very intense periods of water deficit (Carter and Sheaffer, 1983; Bell *et al*., 2007). Nevertheless, field observations have noticed the lack of alfalfa recruitment in these highly eroded slopes, although some alfalfa seeds can actually be found on the soil seed bank (Espigares *et al*., in press). Indeed, our results suggest that field conditions related to water availability in the eroded slopes show important restrictions for the germination of alfalfa seeds, possibly limiting plant recruitment. Furthermore, these conditions could limit more drastically natural colonization by other species, since successful natural colonizing species of human-made slopes of Mediterranean-dry Spain generally show water potential-based critical thresholds for seed germination (between -0.05 and -0.35 MPa; Bochet *et al*., 2007) markedly more restrictive than that we obtained for alfalfa seeds (between -0.20 and -0.60 MPa). In fact, previous works have demonstrated that plant recruitment and natural colonization in these highly eroded coal-mining slopes is seriously constrained by the accumulated impact of water scarcity in seedling emergence and survival and seed production, leading to a species-poor community dominated by some aged drought-tolerant alfalfa plants (Moreno-de las Heras *et al*., 2008; Espigares *et al*., in press). Accordingly, several works carried out in other eroded landscapes of Mediterranean Spain have documented a very acute species loss conducting to very impoverished plant communities, generally dominated by few species specially drought-resistant (Guardia, 1995; Guerrero-Campo and Montserrat-Martí, 2004). The spatial organization of these simple alfalfa communities is conditioned by the consolidation of the interrill-rill pattern of soil moisture distribution. In the most eroded conditions, we observed a preferential distribution of alfalfa plants close to the edge of interrill areas, where they have to face a less intense water stress (probably thanks to the deep intake of water resources around rill beds). Simultaneously, plants in these microsites are safe from the mechanical disturbance associated to concentrated water flows. The emergence of such vegetation pattern, formed by downward spots and stripes adjacent to the flow channelling forms, has been explained in water-restricted environments by the long-term interaction of the patterns of plant growth and senescence with the spatial distribution of both soil moisture and mechanical disturbance (Puigdefábregas and Sanchez, 1996; Saco e*t al*., 2007). As Puigdefábregas *et al*. (1999) asserted, this vegetation pattern shows an exceptional incapacity to control runoff and sediment flows, probably reinforcing the degradation trend through the intensification of the loss of water and soil resources from the slope.

The configuration of this plant distribution pattern disappears in the medium eroded slopes (slopes 3 and 4; rill erosion rate 8-17 t ha^{-1} year^{-1}), where rill networks are spatially discontinuous and soil moisture differences between geomorphic units (rills and interrills) are attenuated. In these slopes, the presence of spotted splays where rills break off could influence vegetation structure and distribution, since this kind of flow discontinuities provides localized areas where vegetation can be favoured by the accumulation of water and soil resources (Wainwright *et al*., 2002). Indeed, several perennial grasses (*Lolium perenne*, *Elymus hispidus* and *Dactylis glomerata*) are widely established in these areas where runoff is discharged and sedimentation occurs (Merino-Martín, 2007), limiting the dominance of alfalfa plants in vegetation composition (*M. sativa* relative abundance: 20-40%; Table 6.1) and deleting the downward aligned pattern of plant distribution.

Both types of plant communities (*M. sativa* type and the more complex ones dominated by perennial grasses) are linked to the gradient of vegetation structure simplification promoted by rill erosion processes acting in the studied reclaimed slopes since their construction. In this way, the impact of such processes in plant community attributes (diversity and biomass) showed a distinctive non-linear (exponential negative) trend, as could be expected given the previous insights on the ecological effects of soil erosion (Thornes, 1990). The non-linear nature of these relationships give notice of the presence of a possible erosion-related characteristic degradation threshold; this threshold in the studied conditions might exist at a rill erosion rate close to 20 t ha^{-1} $year^{-1}$, defining the transition for the development of the low productive and poor *M. sativa* communities spatially organized by the geomorphic distribution of soil moisture and mechanical disturbance.

Conclusions

We can conclude that the development of soil erosion processes with rill formation has a major role on the ecohydrological relations of the studied Mediterranean-dry reclaimed slope systems, by conditioning the availability and spatial distribution of water resources.

The development of rill networks leads to maximize water loss from the slope system by surface runoff. Furthermore, the integration of these runoff routing systems consolidates the emergence of a soil moisture spatial pattern ruled by the distribution of geomorphic forms (rill and interrill units). Thus, when dense rill networks are developed soil moisture penetration on interrill areas is very limited, while soil moisture contents are higher on rill units, whereby concentrated water flows are routed.

As a result of the loss of water resources, drought stress increases in growing vegetation and in addition, unfavourable conditions for plant recruitment and colonization occurs, leading to an exponential decline of species richness and biomass at the slope scale level. On the long term when dense rill networks are developed, soil erosion consequences in the studied environment result in a low productive and species-poor community dominated by some aged *M. sativa* plants, a sown perennial legume specially drought-resistant. These alfalfa communities are spatially arranged by the geomorphic pattern of soil moisture and mechanical disturbance distribution, configuring downward aligned spots adjacent to the channelling forms which simultaneously minimise water-stress and mechanical disturbance.

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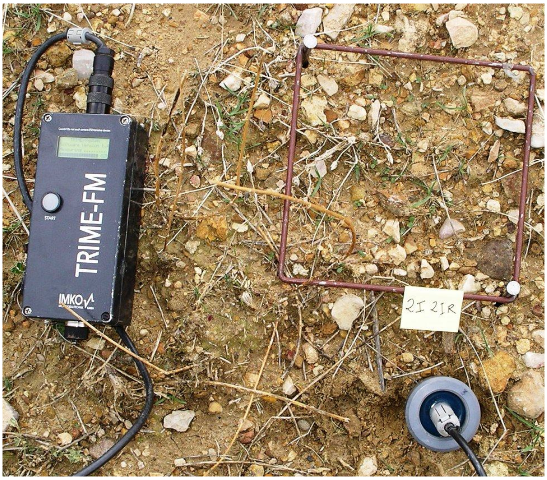
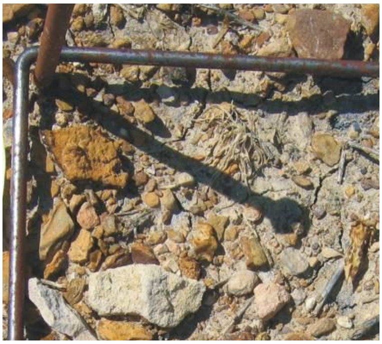
Capítulo 7

Capítulo 7

Dinámica vegetal en laderas restauradas mineras afectadas por procesos de erosión hídrica superficial

Este capítulo reproduce el texto del siguiente manuscrito:

Espigares, T., Moreno-de las Heras, M., Nicolau, J.M. En prensa. Performance of vegetation in reclaimed slopes affected by erosion. *Restoration Ecology*.

Resumen

La colonización y el desarrollo de la vegetación pueden verse afectados por los procesos de erosión hídrica superficial en los ambientes restaurados mineros, condicionando así el éxito de las restauraciones. A pesar de su importancia, existe poca información cuantitativa sobre la magnitud de estos efectos. Durante un ciclo de crecimiento vegetal (2003-04) se ha analizado la dinámica de la vegetación (densidad del banco de semillas, tasas de emergencia y de mortalidad de plántulas, y producción de semillas) en tres laderas restauradas mineras (cuenca minera de Utrillas, España) sometidas a procesos de erosión de diferente intensidad desde su construcción. En la ladera más erosionada se ha observado una baja disponibilidad de agua en el suelo para las plantas (especialmente en los inter-regueros), limitando la emergencia y supervivencia de plántulas, así como la producción de semillas. Los niveles de colonización vegetal observados en las laderas analizadas fueron considerablemente bajos con tasas de erosión en regueros superiores a 17 t ha^{-1} $año^{-1}$ y niveles de desarrollo de la cubierta vegetal inferiores al 30%, especialmente cuando son comparados con los niveles de colonización observados en laderas con niveles de cubierta superiores al 60% y ausencia de procesos de erosión en regueros. Los resultados obtenidos señalan que los procesos de colonización natural bajo las condiciones de estudio (laderas artificiales en clima mediterráneo-continental) pueden verse bloqueados a causa del desarrollo a largo plazo de procesos de erosión con formación de regueros, incrementando las pérdidas de agua de las laderas en forma de escorrentía superficial. En este sentido, tan sólo una profunda intervención humana diseñada para limitar la pérdida de recursos hídricos del sistema, posibilitaría la colonización y el desarrollo de la vegetación en estas laderas intensamente erosionadas.

Palabras clave: ambiente mediterráneo, disponibilidad de agua, erosión en regueros, escorrentía, laderas artificiales, restauración minera.

Performance of vegetation in reclaimed slopes affected by soil erosion

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Abstract

Soil erosion in reclaimed mines may affect plant colonization and performance, and may compromise restoration success; however, the magnitude of this effect has seldom been quantified. We monitored the dynamics of vegetation (seed bank density, seedling emergence, plant mortality, and seed production) during a growing season (2003-2004) in three constructed slopes with differing past erosion rates. The slopes are located in the Utrillas coalfield in Spain, which experiences a Mediterranean-Continental climate. In the most eroded slope, soil water availability was lower -especially in the interrill areas- and seedling emergence rate, plant survival, and seed production were also significantly lower than on the less eroded slopes. We found that vegetation recovery is dramatically constrained when rill erosion rate is 17 t ha^{-1} $year^{-1}$ and plant cover is 30%, but this effect disappears when plant cover is higher than 60%. Soil erosion in constructed slopes appears to inhibit natural plant colonization processes by increasing runoff water loss over the long-term. Thus, when rill erosion networks develop, human intervention would be needed to minimize the loss of water and facilitate vegetation colonization.

Key words: constructed slopes, Mediterranean environment, mining reclamation, overland flow, rill erosion, water availability.

Introduction

Soil erosion is one of the most significant barriers to the success of restoration practices in constructed slopes derived from mining activities (Whisenant, 2005). Many failures in successful restoration of mining areas are the result of soil erosion processes triggered by an excess of overland flow on structurally poor soils with generally low infiltration rates (Nicolau and Asensio, 2000).

Traditionally, the role of soil erosion on ecosystems has been interpreted as an abiotic exploitation agent, responsible for the loss of nutrients (Lü *et al*., 2007) and propagules (Cerdà and García-Fayos, 1997; Chambers, 2000). Soil erosion may reduce water availability by reducing soil depth and contributing to the formation of surface soil crusts (Pimentel *et al*., 1995; Sarah, 2004). If soil erosion promotes the development of rill networks, water availability may be further reduced, since rills increase runoff connectivity on the slopes and provide efficient pathways to drive water out of the system (Favis-Mortlock *et al*., 2000; Bracken and Croke, 2007). In this way, erosion can negatively affect plant colonization and performance by reducing the availability of seeds, nutrients and water in soil.

On the other hand, it is well known that vegetation cover reduces soil erosion (Elwell and Stocking, 1976; Francis and Thornes, 1990; Bochet and García-Fayos, 2004) by decreasing soil erodibility, effective precipitation, and kinetic energy of runoff and raindrops (Brandt, 1989; Domingo *et al*., 1998; Martínez-Mena *et al*., 2000). Thus, one of the primary targets in the reclamation of artificial slopes is to reach optimal vegetative cover for the prevention of soil erosion (Redente and Depuit, 1988; Andrés and Jorba, 2000).

The relationships between erosion and vegetation are complex since the soil-vegetation system has feedback mechanisms which regulate soil formation, plant development, and erosion-sedimentation processes (Kirkby *et al*., 1998; Puigdefábregas *et al*., 1999). However, more attention has been paid to the effects of vegetation on erosion than the reverse (García-Fayos and Cerdà, 1997). During the last decade, many studies have tried to identify the limiting factors controlling plant colonization in highly eroded areas of Mediterranean environments subjected to seasonal drought. García-Fayos *et al*. (1995) found that the lack of vegetation in badlands was not caused by seed removal as few seeds were lost by erosion (<13%), and seed rain was always greater than seed outputs. It is also known that the hardness of soil crusts adversely affects plant establishment by interfering with seedling emergence and reducing the infiltration rate, causing loss of water through runoff (Awadhwal and Thierstein, 1985). Various studies suggest that the primary factor controlling plant colonization in these areas is the effect of water availability on germination and plant establishment (García-Fayos *et al*., 2000; Bochet *et al*., 2007).

Water availability is certainly one of the primary limiting factors affecting performance of vegetation in reclaimed slopes, especially in areas subjected to water stress such as Mediterranean environments (Martínez-Ruiz and Marrs, 2007). Several authors have documented that high mortality rates of seedlings are related to water stress in artificial slopes created by mining activity (Bell and Ungar, 1981). Also, studies on ecological succession in mine reclaimed slopes have identified water availability as a major driver of successional trajectories (Wiegleb and Felinks, 2001).

In reclaimed mining areas of Teruel province (Spain), Nicolau (2002) evaluated some mechanisms by which soil erosion can reduce water availability in eroded artificial slopes, namely: reduction of total soil depth and soil surface roughness, crust formation, and soil structure degradation. In this study, the hydrological response of reclaimed slopes was monitored 3 and 7 years after their construction, and substantial differences were observed: slopes in which rills formed in the early phase of slope development lost their functionality as ecological succession proceeded, and slopes in which the density of rills increased experienced high losses of water by surface runoff and lower rates of plant colonization. It can therefore be concluded that rill development in artificial slopes may limit water storage in soils and negatively affect plant colonization and the establishment of protective cover. Indeed, the loss of water resources has been associated with the initiation of feedback loops in both natural and reclaimed water-restricted environments, which causes sharp changes in ecosystem structure and functionality (Moreno-de las Heras *et al*., 2008; Turnbull *et al*., 2008).

In the present study we analyze the performance of vegetation throughout a growing season in three reclaimed slopes of the opencast coal mining area of Teruel that have been subjected to different soil erosion intensities since their construction. The slopes were reclaimed in 1988-89, with the same substrate, orientation, and general restoration treatments, but they differ in the density of rills as a result of failures in their geomorphological design. We expected that soil erosion processes would affect plant performance negatively in terms of the rate of seedling emergence, plant mortality, and seed production. We also expected that plant performance would be less successful in the slopes with higher density of rills, corresponding with lower soil water availability. In a previous study in the area (Moreno-de las Heras *et al*., 2008), we identified rill erosion as a driving force in plant succession on constructed slopes, leading to plant communities with low diversity and cover. The present study intends to identify the mechanisms through which soil erosion affects plant communities in reclaimed slopes. Halle and Fattorini (2004) emphasized the need to analyze the processes of germination, growth, and reproduction of plants in restored sites to understand the underlying processes involved. Our study follows this path, and can contribute to the understanding of the ecological effects of soil erosion in reclaimed ecosystems.

Methods

Study site

The study was carried out in the "Utrillas fieldsite", located in the mine El Moral, in the Utrillas coalfield, central-eastern Spain (lat 40º47'24'' N, long 0º49'28'' W).

This area is situated in the Iberian chain with a mean altitude of 1100 m.a.s.l. Mean annual air temperature is 11ºC (mean monthly temperature 6.8ºC in December and 23.5ºC in July), and the air frost period runs from October to April. The climate is Mediterranean-Continental and the local moisture regime is Mediterranean-dry, according to Papadakis (1966). The rainy period primarily occurs in spring and autumn, with a mean annual precipitation of 466 mm. Potential evapotranspiration is 758 mm, with a hydrological deficit of 292 mm between June and October. The mean of the annual rainfall events is approximately 50 mm, which includes some convective rainstorms of high intensity, particularly in summer (Peña *et al*., 2002).

Mining activities in the area were completed in 1988-89, at which time the company Minas y Ferrocarril de Utrillas S.A. undertook restoration practices. We selected three adjacent artificial North-facing slopes*, with a rectilinear shape and 20º slope angle. All slopes received the same reclamation treatments, consisting of a 1 m superficial layer of overburden substrate revegetated by seeding a mixture of perennial grasses and leguminous herbs (*Festuca rubra*, *F. arundinacea*, *Poa pratensis*, *Lolium perenne*, *Medicago sativa* and *Onobrychis viciifolia*; nomenclature follows Tutin *et al*., 1964-1980), with a sowing density of 350-400 kg seeds/ha. The overburden substratum came from the Escucha Cretacic formation of Albian age; this is a non-sodic and clay-loam textured substratum with a basic pH (Table 7.1). In spite of having very similar initial features, monitoring of the slopes 2 years after their construction showed differences in the intensity of soil erosion influenced by the accumulation of overland flow in water-contributing areas upslope, as a result of design flaws (Nicolau 1996, 2002). No replicated slopes for a given erosion

Table 7.1. Basic characteristics (soil, surface traits, and hydrological features) of the three experimental slopes. Mean ± standard deviation values are shown.

	N	Slope 2	Slope 3	Slope 5
Topography				
Slope length (m)		50	75	60
Slope gradient (°)		20	20	20
Length of water-contributing area (m)		8.0	6.5	na
Soil traits				
Stoniness (%)	15	27.4 ± 3.5^{a}	26.2 ± 4.1^{a}	24.5 ± 3.3^{a}
Sand (%)	15	33.5 ± 3.7^{a}	33.8 ± 3.0^{a}	36.3 ± 2.7^{a}
Silt (%)	15	33.8 ± 1.6^{a}	30.8 ± 1.8^{ab}	26.6 ± 4.5^{b}
Clay (%)	15	32.8 ± 2.9^{a}	35.4 ± 2.1^{a}	37.1 ± 2.9^{a}
Texture	15	Clay loam	Clay loam	Clay loam
WHC (%, w/w)	15	11.9 ± 0.2^{a}	10.7 ± 0.5^{a}	10.6 ± 1.5^{a}
pH -H_2O; w/v: ½-	15	8.0 ± 0.1^{a}	8.0 ± 0.1^{a}	7.9 ± 0.1^{a}
EC -w/v: ½- (dS m^{-1})	15	0.26 ± 0.14^{a}	0.20 ± 0.10^{a}	0.23 ± 0.03^{a}
CEC ($cmol_c$ kg^{-1})	15	23.3 ± 3.3^{a}	28.3 ± 1.0^{a}	22.1 ± 3.5^{a}
ESP (%)	15	0.27 ± 0.11^{a}	0.13 ± 0.01^{a}	0.18 ± 0.03^{a}
Bulk density (g cm^{-3})	45	1.49 ± 0.12^{a}	1.39 ± 0.17^{a}	1.23 ± 0.17^{b}
Organic matter (%)	15	0.56 ± 0.23^{a}	1.27 ± 0.35^{a}	2.00 ± 0.74^{b}
Surface traits				
Vegetation cover (%)	42	9.9 ± 12.2^{a}	26.2 ± 23.8^{a}	59.4 ± 17.6^{b}
Aboveground biomass (g m^{-2})	42	32.5 ± 43.7^{a}	56.9 ± 54.7^{a}	205.4 ± 96.4^{b}
Hydrological features				
Runoff coefficient (%)		21.0	15.9	4.5
Rill density (m m^{-2})		0.78	0.58	0.00
Rill erosion rate (t ha^{-1} $year^{-1}$)		45.0	16.9	0.0

Abbreviations: N: number of samples; WHC: water holding capacity; w/w: ratio of water (weight) / soil (weight); EC: electrical conductivity; w/v: ratio of soil (weight) / water (volume); CEC: cation exchange capacity; ESP: exchangeable sodium percentage; na: not applicable.

Different letters (a-b) within rows indicate differences at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney.

*En la versión publicada de este artículo las laderas experimentales aparecen identificadas con otros códigos (slopes 1, 2 and 3). Los códigos han sido substituidos por otros (slopes 2, 3 and 5) para hacerlos coincidir con los utilizados en el resto de los capítulos que integran esta tesis.

level were available. Thus, we considered subplots within each slope as replicates.

Data sampling for characterization of vegetation, soil and hydrology of slopes

In order to characterize soil traits, five composite soil samples were collected from the top 10 cm in each experimental slope (each formed by six subsamples distributed at random, which were homogeneously mixed in an attempt to minimise spatial variability). Soil samples were air dried and sieved (2 mm sieve). General physico-chemical characteristics were determined using the standardised methods proposed by the Spanish Ministry of Agriculture (MAPA 1994). Soil water retention at both field capacity and the willing point (-0.03 and -1.50 MPa respectively) was determined following the pressure chamber procedure (Klute 1986). Water holding capacity was subsequently calculated as the difference between these values (Descheemaeker *et al*., 2006). Finally, soil bulk density was determined from 15 unaltered soil sample cores (3 cm height by 5 cm diameter) collected at random in each slope.

Vegetation sampling took place in June 2004. In each slope, vegetation cover was quantified in fourteen 50 x 50-cm quadrats, distributed at random. Aboveground biomass was collected in these quadrats, and subsequently oven-dried (60ºC; 72 h) and weighed.

To assess soil erosion processes, rill density was measured in each slope (m of linear rill/ m^2 of area). Rill sections (width and depth) were directly determined in all rills intercepted by three equidistant cross-slope transects of 30 m length. Historical averaged rill erosion rate (t ha^{-1} $year^{-1}$) in each slope was quantified using the former rill network dimensions as well as the bulk soil density and slope age from reclamation (Morgan 1995).

Cumulative runoff rate in the analysed experimental slopes is provided for a later hydrological year (from October 2005 to October 2006). This data was obtained in 3 x 15-m runoff plots (Moreno-de las Heras *et al*., 2007).

Experimental design

We established ten 5 x 5-m plots randomly distributed in two positions on each slope: 5 upslope (upper half of the slope) and 5 downslope. In slopes 2 and 3, with rills already developed, we compared the dynamics of vegetation between rill and interrill sites. It must be stressed that rill microsites have very little vegetative cover (< 2%) due to the mechanical perturbations provoked by water flow. Slope 5 was considered an interrill area as it had no rills.

Soil moisture

Volumetric soil moisture content (%) in the upper 15 cm of soil was measured with TDR technology (Topp *et al*., 1980; Topp and Davis, 1985) by using a TRIME FM (Imko®) instrument. Soil moisture was sampled during vegetation data collection beside each permanent plot established for the monitoring of plant mortality.

Soil seed bank composition and seedling emergence

In September 2003, before the arrival of autumn rains, we collected soil samples (60 cm^2 x 4 cm depth) to analyze the composition of the soil seed banks. In each plot we collected 8 soil samples located at random, 4 in rills and 4 in interrills. Each of these samples was subdivided into 4 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.

In November 2003, after the conclusion of autumn germination in the field, we measured the number of emerged seedlings in 20 x 20-cm quadrats randomly distributed in each plot, 3 in rills and 3 in interrills. Relative emergence rate for each plot was calculated by dividing the mean number of emergences recorded in the field by the density of germinable seeds in soil obtained in the greenhouse experiment.

Plant mortality

In October 2003 we established 8 permanent 20 x 20-cm quadrats randomly distributed in each plot, 4 on rill and 4 on interrill areas to monitor seedling survival. Quadrats on bare soil were discarded, which may have importance for the interpretation of the results in rills where plant cover is very scarce and individuals concentrate in few safe-sites. We recorded the performance of each seedling emerging from October 2003 to June 2004 twice a month in autumn and spring, and once a month in winter. During each sampling day we recorded every seedling failure and the cause of mortality. We used orthogonal photographs of each quadrat 1.30 m above ground (Andres and Jorba, 2000) to complement field observations. This permitted us to have more observational data of seedlings in order to better assign the possible cause of death. We differentiated four causes of mortality: (i) frost (seedlings were dessicated, usually after becoming reddish, during winter months), (ii) soil erosion (including seedlings that were removed with sediments and seedlings that were buried under sediments and did not re-emerge), (iii) sheep-trampling, and (iv) drought (seedlings dessicated in late spring, coinciding with the period of minimum water availability). Other factors may have influenced seedling mortality, although field observations suggested that these were not major causes of death. We calculated total and plot-specific mortality rate by dividing the number of dead seedlings by the total number of emerged seedlings in each quadrat. We also estimated the relative percentage of seeds in soil that became adult plants by multiplying the emergence rate of seedlings by their survival rate in each plot.

Seed production

We analyzed the influence of soil erosion on seed production and seed weight. For this purpose we chose the species *Aegilops geniculata*, as it is one of the most conspicuous in the studied slopes (Table 7.2). In summer 2004, after the seeds had matured, we collected all *A. geniculata* caryopses in 50 x 50-cm quadrats, 12 in the upper part and 12 in the lower part of each slope (and within each part, 6 in the rills and 6 in the interrills, also randomly distributed). We separated all the seeds and weighed them after air-drying. To explore the differences in the weight of produced seeds between slopes, we differentiated between light (<0.005g) and heavy seeds (>0.005g), as it is well known that *Aegilops* species produce two types of seeds within the same plant progeny that differ in mass and germinability (Marañón, 1989).

Data analysis

We used nonparametric statistical tests (Kruskall-Wallis and Mann-Whitney U tests) to explore the differences between the three slopes in the soil properties, the density of soil seed banks, seedling emergence rate, relative percentage of seeds in soil that became adult plants, and *A. geniculata* seed production and average seed weight. Non-parametric tests were selected because data could not be transformed. The use of nonparametric analyses also follows the recommendation of Ruxton and Beauchamp (2008) to use the Kruskal-Wallis test with environmental data: if 'the sample distributions are similar and symmetric' and if 'the variances of [these] ranks are similar for all the groups'. A Non-metric Multidimensional Scaling (NMDS) analysis was used for data on floristic composition of the soil seed bank. The graphic representation of the samples in the ordination space enabled us to measure the heterogeneity in floristic composition of the different slopes. Soil moisture and relative percentage of causes of mortality were transformed (arcsine (square root(x)) to achieve normality. Soil moisture data from the different sampling dates were analyzed with repeated measures ANOVAs (with time as within subjects factor and slope as between subjects factor). A MANOVA test was used to analyze the differences between slopes in the relative percentage of causes of mortality, using the plot as experimental unit. Means were back-transformed to report the data. Multivariate analysis was performed with PC-ord package (McCune and Mefford, 1997); the STATISTICA package was used for the remaining statistical analyses (Statsoft Inc., 1996).

Results

Soil properties

Although general chemical properties (pH, conductivity, cation exchangeable capacity, and exchangeable sodium percentage), stoniness, texture, and the water holding capacity of the soils were rather homogeneous between the three analysed slopes (Table 7.1), some important differences regarding soil bulk density (Kruskall-Wallis H = 14.74, N = 45, df = 2, $p<0.001$) and organic matter content (Kruskall-

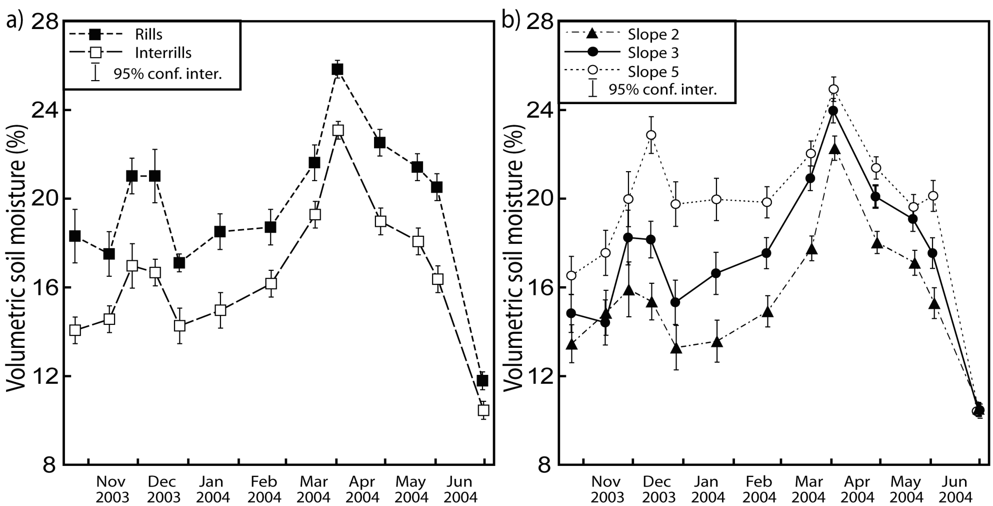

Figure 7.1. Mean volumetric soil water content and 95% confidence interval in the different sampling dates along the growing season in a) rills and interrills and b) interrills of the three slopes.

Wallis H = 9.92, N = 15, df = 2, p=0.007) were detected. Soil organic matter in slopes 2 and 3 was significantly lower than in slope 5 (Table 7.1). Additionally, soil bulk density in slopes 2 and 3 was significantly higher than in slope 5 (Table 7.1), where a high density of herbaceous roots was observed.

Soil moisture

More water was available in the rills than in the interrills in slopes 2 and 3 (Repeated Measures ANOVA, $F_{1,60}$ = 270.49, p<0.001; Fig. 7.1a). Due to the differences in hydrology between rills and interrills, we analysed the water content of the soils from both sites separately. Results of the Repeated Measures ANOVA with the data on volumetric water content in interrills revealed significant differences between slopes: slope 2 had the lowest values while the uneroded slope (slope 5) had the highest ($F_{2,42}$ = 107.64, p<0.001; Fig. 7.1b). Rills in slope 3 had more water than in slope 2, however ($F_{1,28}$ = 107.50, p<0.001). Slope position had no effect in the water content.

Soil seed banks and seedling emergence

Twenty-five different species were identified in the soil seed banks of the three slopes (Table 7.2). The NMDS with the floristic composition data (final stress 0.24) showed a greater heterogeneity between samples in slopes 2 and 3 on the basis of Euclidean distances (Fig. 7.2) compared with samples in the uneroded slope (slope 5). We detected an effect of slope position on floristic composition in slope 5, with significant differences occurring between the axis 2 coordinates of the samples from the upper and lower part of this slope (Mann-Whitney U test, Z= -2,61, p=0.009).

There were significant differences in the total density of seeds between the three slopes

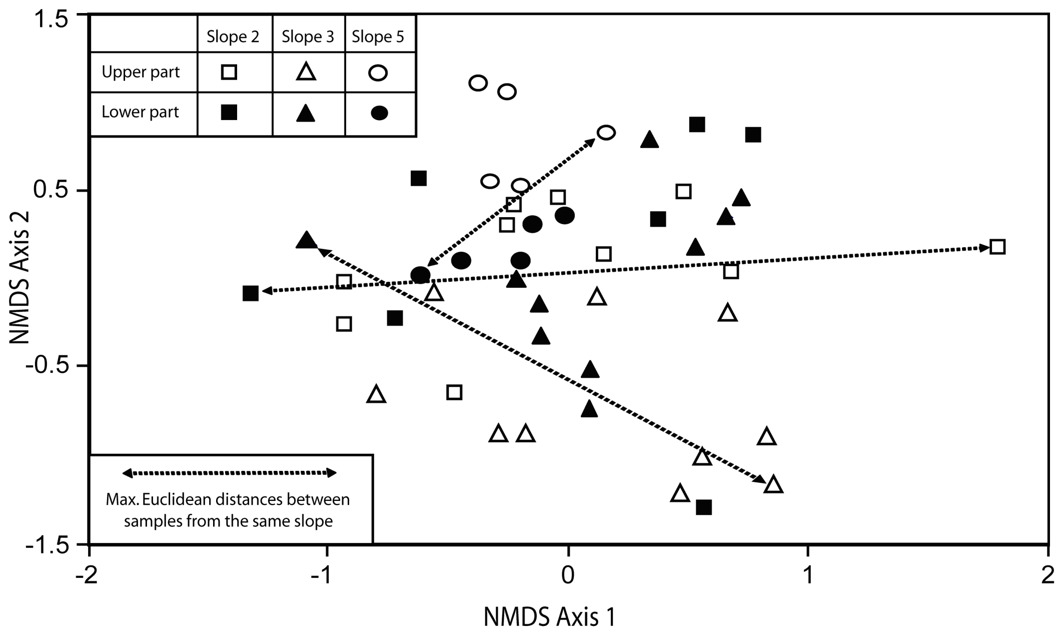

Figure 7.2. NMDS ordination graph with the floristic composition of the soil seed bank data.

Table 7.2. Mean (± SE) density of seeds in soil (seeds m^{-2}) of each of the identified species in each slope and erosion area (R: rills, IR: interrills). Means (± SE) of total seed density in soil (seeds m^{-2}), emerged seedling density (seedlings m^{-2}), seedling emergence rate in the field (%), total plant mortality (%), and relative percentage of seeds in soil that became adult plants (%) are also indicated.

	Slope 2		Slope 3		Slope 5
Species	R	IR	R	IR	IR
Aegilops geniculata	86.3 ± 35.4	16.4 ± 9.1	12.3 ± 12.3	49.3 ± 14.8	542.8 ± 126.5
Androsace maxima	0	0	0	0	4.1 ± 4.1
Anthemis tuberculata	0	0	4.1 ± 4.1	0	4.1 ± 4.1
Brachypodium retusum	0	0	16.4 ± 16.4	0	333.1 ± 234.2
Bromus hordeaceus	0	0	0	0	24.7 ± 24.7
Bromus rubens	78.1 ± 53.6	4.1 ± 4.1	49.3 ± 49.3	20.6 ± 9.2	53.5 ± 29.4
Bromus tectorum	0	0	0	0	4.1 ± 4.1
Desmazeria rigida	4.1 ± 4.1	0	20.6 ± 14.1	0	271.4 ± 104.6
Echinaria sp.	4.1 ± 4.1	4.1 ± 4.1	4.1 ± 4.1	0	0
Echium sp.	4.1 ± 4.1	0	0	0	0
Filago vulgaris	0	0	61.7 ± 37.9	24.7 ± 9.1	24.7 ± 16.4
Hieracium pilosella	0	0	0	0	4.1 ± 4.1
Hordeum murinum	20.6 ± 11.1	4.1 ± 4.1	4.1 ± 4.1	0	0
Juncus sp.	0	0	0	8.2 ± 8.2	0
Lamium amplexicaule	0		0	4.1 ± 4.1	4.1 ± 4.1
Lolium sp.	0	0	4.1 ± 4.1	0	4.1 ± 4.1
Medicago sativa	41.1 ± 33.0	4.1 ± 4.1	4.1 ± 4.1	4.1 ± 4.1	0
Papaver rhoeas	0	0	0	4.1 ± 4.1	4.1 ± 4.1
Plantago lanceolata	0	0	8.2 ± 8.2	4.1 ± 4.1	12.3 ± 8.8
Sanguisorba minor	0	0	4.1 ± 4.1	0	4.1 ± 4.1
Santolina chamecyparissus	0	0	78.1 ± 36.0	16.4 ± 9.1	4.1 ± 4.1
Senecio vulgaris	0		0	4.1 ± 4.1	4.1 ± 4.1
Sisymbrium orientale	24.7 ± 12.6	8.2 ± 8.2	24.7 ± 9.1	32.9 ± 12.0	4.1 ± 4.1
Thymus vulgaris	0	0	0	0	16.4 ± 12.6
Xeranthemum annum	0	0	41.1 ± 41.1	4.1 ± 4.1	123.4 ± 88.2
Undetermined graminoids	45.2 ± 15.6	24.7 ± 11.0	180.9 ± 109.0	37.0 ± 17.8	468.8 ± 176.2
Undetermined forbs	41.1 ± 13.7	12.3 ± 6.3	82.2 ± 39.2	32.9 ± 10.2	106.9 ± 26.1
Total seeds in soil	349.5 ± 102.1^{a}	78.1 ± 21.6^{a}	600.3 ± 170.3^{b}	246.7 ± 31.8^{b}	2023.0 ± 265.3^{c}
Emerged seedling density	34.3 ± 14.3^{a}	14.4 ± 4.9^{a}	170.8 ± 29.8^{b}	149.2 ± 27.3^{b}	1616.2 ± 146.3^{c}
Seedling emergence rate	16.9 ± 11.2^{a}	13.0 ± 4.9^{a}	75.7 ± 37.7^{b}	67.1 ± 13.7^{b}	90.0 ± 11.3^{c}
Total plant mortality	23.1 ± 5.7^{a}	76.6 ± 6.8^{b}	22.0 ± 4.8^{a}	70.0 ± 5.9^{b}	6.01 ± 1.7^{c}
% Seeds became adult plants	12.7 ± 8.3^{a}	2.7 ± 1.0^{a}	58.5 ± 28.7^{b}	22.3 ± 6.4^{b}	84.5 ± 10.6^{c}

Nomenclature follows Tutin *et al.* (1964-1980).
Different letters (a-c) indicate significant differences between slopes at α=0.01. Tested using Mann-Whitney U test.

(Kruskall-Wallis H = 26.73, N = 50, df = 2, p < 0.001), with more seeds in slope 5 than in the eroded slopes (Table 7.2). Slope position did not affect seed density. Results of the evaluation of erosion zone data indicated that in slope 2 there were more seeds in the rills than in the interrills (Mann-Whitney U test, Z= 2.60, p = 0.008). Differences in seed density between rills and interrills in slope 3 followed the same pattern but with low statistical significance (p = 0.057).

Seedling emergence rate in the field was lower in the more eroded slope and reached the highest value in slope 5 (Kruskall-Wallis H = 22.66, N = 50, df = 2, p<0.001; Table 7.2). No significant effect of slope position or erosion zone (rills and interrills) on seedling emergence rate was found.

Plant mortality

Total plant mortality during the whole growing season was higher in the slopes subjected to higher soil erosion ($F_{2,74}$ = 10.56, p<0.001; Table 7.2). No significant effect of slope position was detected. In slopes 2 and 3 total plant mortality was greater in interrills than in rills ($F_{1,56}$ = 67.6, p<0.001; Table 7.2). The MANOVA test with the relative percentages of mortality causes showed significant differences between slopes ($F_{8,148}$ = 3.29, p=0.001; Fig. 7.3). Drought was the greatest cause of mortality in all slopes (42.1% ± 4.7, mean ± SE) followed by erosion (15.5% ± 3.6, mean ± SE) and frost (14.6% ± 2.9, mean ± SE). Erosion affected only slopes 2 and 3. Plant mortality from trampling by sheep was almost negligible (0.1% ± 0.0, mean ± SE) and only appeared in slopes 3 and 5 (Fig. 7.3). The relative percentage of seeds in soil that became adult plants was significantly higher in slope 5 (Kruskall-Wallis H = 27.17, N = 47, df = 2, p<0.001; Table 7.2), and no effect of erosion zone (rill and interrill) or slope position was detected.

Seed production and seed weight

We found a significantly higher density of *Aegilops geniculata* seeds in slope 5 (Table 7.3); no effect of slope position or erosion zone was detected. Both heavy and light seeds from the more eroded slope (slope 2) had a significantly lower weight than the seeds produced in slopes 3 and 5 (Table 7.3). No effect of slope position or erosion zone (rill and interrill) was detected on seed weight.

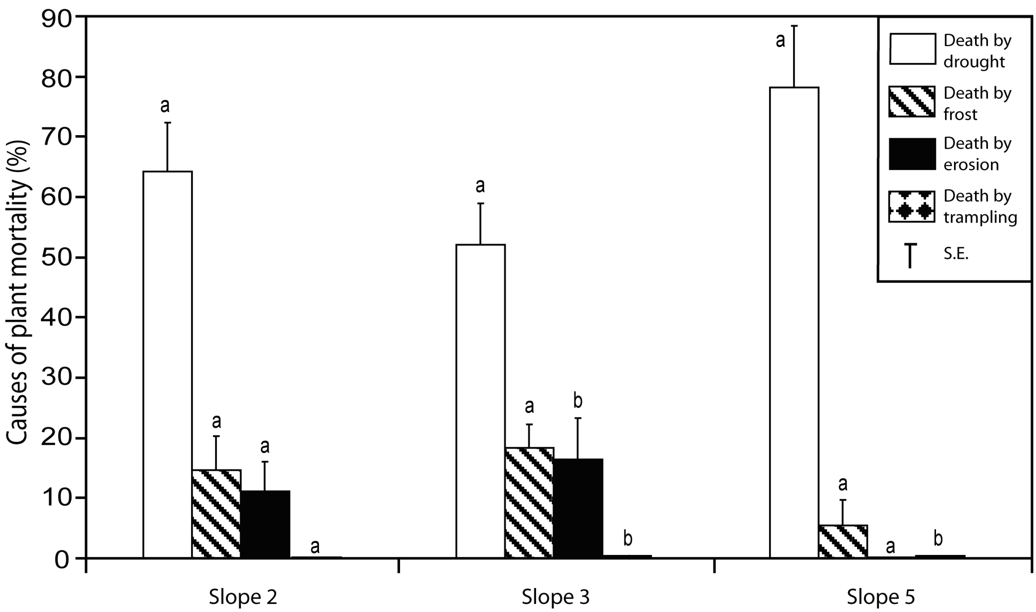

Figure 7.3. Relative percentage of the different causes of plant mortality in the three slopes. Different letters indicate significant differences between slopes (LSD test, p<0.05).

Table 7.3. Mean number (± SE) of *Aegilops geniculata* seeds produced (seeds m^{-2}), and mean weight (± SE) of the two types of *A. geniculata* seeds in each slope.

	Slope 2	Slope 3	Slope 5	Kruskal-Wallis (H)	N	*p*<
Seeds produced (seeds m^{-2})	56.6 ± 14.7^{a}	184.7 ± 47.6^{ab}	887.6 ± 382.4^{b}	13.35	62	0.001
Light seed weight (mg)	3.3 ± 0.08^{a}	3.7 ± 0.11^{b}	3.6 ± 0.04^{b}	14.58	892	0.001
Heavy seed weight (mg)	8.6 ± 0.18^{a}	10.9 ± 0.26^{b}	10.3 ± 0.15^{b}	48.83	2855	0.001

Different letters (a-b) indicate significant differences between slopes.

Discussion

Monitoring of vegetation dynamics in mine reclaimed slopes subjected to different soil erosion intensity during 16 years reveals that reduced plant performance is associated with increasing soil erosion rates. All development stages measured: seedling emergence rate, plant survival, seed production, and seed weight, were negatively affected as rill erosion increased.

Soil seed density was highly variable between slopes (ranging from 78 to 2023 seeds m^{-2}), with the lowest values found in the most eroded slope. These values are comparable to the low seed densities found in natural badlands of Mediterranean Spain (García-Fayos *et al*., 1995, 2000). The differences in soil seed densities clearly reflect the variation in vegetation cover between slopes. In contrast, density of emerged seedlings varied much more between slopes than seed densities, ranging from 14 to 1616 seedlings m^{-2}. Although seed densities were similar to other degraded areas, the rate of seedling emergence in the uneroded slope was higher: the maximum seedling emergence density found by García-Fayos *et al*. (2000) in Spanish badlands was 273 seedlings m^{-2}; and Elmarsdottir *et al*. (2003) found values around 300 seedlings m^{-2} in reclaimed slopes of Iceland. Therefore, higher soil erosion rates imply a reduction in seedling emergence.

Soil erosion also affected the spatial heterogeneity of soil seed banks, as more seeds were found in the rills than in the interrills, possibly because rills trap seeds in the eroded slopes (Tsuyuzaki *et al*., 1997; Chambers, 2000).

We observed higher mortality in the eroded slopes, primarily due to drought. Hence, plant establishment is impeded in reclaimed slopes affected by soil erosion. Indeed, while an average of 84% of the seeds in soil become adult plants in slope 5, only 9% do so in slope 2. García-Fayos *et al*. (2000) found that plant mortality in Spanish badlands reached values up to 98%, and was mostly related to drought. In our reclaimed slopes, plant mortality ranged from 77% in interrill areas of the most eroded slope to 6% in the uneroded slope. The lower plant mortality found in rills is probably due to its higher soil moisture compared with interrills; however, this is not representative of the slope overall, because only few safe sites in rills, such as small depressions protected from water flow by rock fragments, contain plants. Mortality directly related to erosion or frost was less common, each affecting about 5% of the plants. García-Fayos *et al*. (2000) found similar values for erosion-related mortality of plants in badlands of SE Spain.

Finally, we have observed that soil erosion affects seed production negatively, as fewer and smaller seeds of the species *Aegilops geniculata* were found in the most eroded slope. Although density of seeds may be influenced by the differences in vegetation cover between slopes, the impact on seed weight must be an indirect consequence of soil erosion effects on soil water availability. The influence of limited resources, mainly water, on seed production, by means of a reduction in seed number or size is well known (Stephenson, 1981; Pyke, 1989). Baalbaki *et al*. (2006) observed that in several *Aegilops* species, seed number and weight were the attributes most affected by severe water stress in Mediterranean semiarid areas of Lebanon.

These negative effects of soil erosion on plant performance were associated with a reduction of the soil water content in the more eroded slopes. This would have constrained plants growing in the eroded slopes to withstand more intensive water stress. This might also explain the internal organization of the vegetation across the slope 5 (the differences in floristic composition between upslope and downslope areas) and its absence on the eroded slopes, where the pattern of rills and interrills would be the primary factor influencing the structure of the plant community.

Our results suggest that long term effects of soil erosion in constructed slopes inhibit vegetation development by limiting water availability. We found lower soil moisture in the slopes with more developed rill networks. In addition to the draining effect of rill networks, feedback mechanisms between vegetation and soil may be involved in these limitations. Indeed, sixteen years of different soil erosion rates have clearly affected soil formation processes on the slopes in the study area and, consequently, water infiltration. This has resulted in limitations to the accumulation of organic matter and the impact of vegetation roots in the bulk density. Overall water availability limitations are also assessed by measured runoff coefficients along the 2005-06 hydrological year (Moreno-de las Heras *et al*., 2007). Although all slopes received the same rainwater (615 mm), the high runoff rates of the eroded slopes may have limited their water availability. The runoff rate of slope 5 was 4.5%, which is similar to the values found in non-degraded natural slopes and successfully reclaimed mining slopes of the Mediterranean Spain (Puigdefábregas *et al*., 1999; Nicolau, 2002; Martínez-Murillo and Ruiz-Sinoga, 2007). Conversely, runoff rates in slopes 2 and 3 ranged from 15 to 20%, indicative of a noticeable loss of water by surface runoff. Other works carried out in Mediterranean Spain found a high similarity between the spontaneous flora in eroded zones and in uneroded areas naturally subjected to severe water stress, which agrees with the hypothesis that plants growing in eroded areas must face a higher water deficit related to soil erosion processes (Guerrero-Campo and Monserrat-Martí, 2004).

The three constructed slopes represent three different points of a soil erosion intensity gradient. The analysis of vegetation dynamics in these slopes allows us to conclude that plant performance is more similar in slopes 2 and 3 than in slope 5. The model of competition for soil moisture between vegetation and soil erosion proposed by Thornes (1985) established instability regions in the erosion-vegetation system depending on critical values of soil erosion rate and vegetation cover. We can assume that slopes 2 and 3 fall into this instability situation, in which thresholds of erosion and vegetation cover have been surpassed and control of soil and water resources rely on erosional rather than vegetative processes. Therefore, threshold values of vegetation cover and erosion rates for our system may be close to those of slope 3: 30% and 17 t ha^{-1}

year^{-1} respectively. Although these values should be evaluated in additional experimental studies, they agree with values proposed by other authors in Mediterranean Spain (Andrés and Jorba, 2000; Francis and Thornes, 1990; Gimeno-García *et al*., 2007).

We must stress the importance of improving the design of constructed slopes to avoid the formation of water contributing areas upslope that concentrate overland flow and may promote soil erosion processes. Hancock and Willgoose (2004) demonstrated that runoff control at the top of reclaimed slopes is crucial to the long-term stability of such structures. This is of particular importance for Mediterranean environments, where, when a threshold is passed, an extensive and costly human intervention is needed, and in some cases, a state of irreversibility is reached (Aronson *et al*., 1993).

We can thus conclude that soil erosion acting in the long term may impose serious limitations on plant performance that are associated with a reduction in water availability. In this sense, soil erosion could act as an ecological filter that may direct the community assembly process towards an irreversible situation in which natural plant colonization is heavily constrained.

Implications for practice

- The early triggering of soil erosion processes in constructed slopes leads to situations in which performance of vegetation is seriously constrained by the loss of water resources.

- To avoid situations in which natural plant colonization is inhibited, vegetation cover of at least 30% and rill erosion rates below 17 t ha^{-1} year^{-1} are required in Mediterranean-Continental reclaimed environments.

- In highly eroded constructed slopes with developed rill networks, human intervention is needed to prevent the loss of water and facilitate vegetation recovery.

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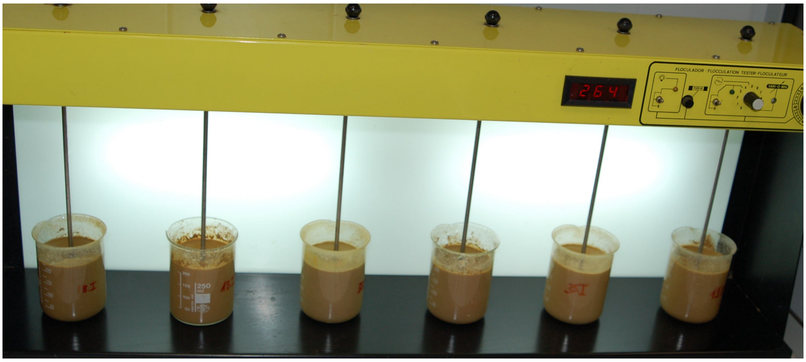
Capítulo 8



Capítulo 8

Desarrollo de la estructura física y la funcionalidad biológica del suelo en estériles mineros afectados por erosión hídrica superficial bajo clima mediterráneo-continental

Este capítulo reproduce el texto del siguiente manuscrito:

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Resumen

Este trabajo se centra en el estudio del desarrollo de la estructura física y funcionalidad biológica del suelo tras 18 años de evolución en estériles restaurados mineros afectados por procesos de erosión de diferente intensidad (tasas de erosión en regueros desde 0 a 70 t ha^{-1} $año^{-1}$). Para abordar este objetivo se determinaron diferentes parámetros: la distribución de tamaños y estabilidad de los agregados del suelo [diámetro medio (MWD), las proporciones de microparticulas y macropartículas (MIP y MAP) y la estabilidad de los agregados (AS)]; el tamaño de la poblaciones microbianas del suelo [carbono de la biomasa microbiana (MBC)] y su actividad [respiración basal (RB) y actividad enzimática de la deshidrogenasa (DHA)]; y la actividad de diferentes hidrolasas del suelo [fosfatasa (PHA), β-glucosidasa (β-GA) y ureasa (UA)]. Los resultados obtenidos a partir de análisis de gradientes (llevado a cabo mediante Análisis de Componentes Principales) señalan a la acumulación en el suelo de materia orgánica derivada de la vegetación restaurada como factor desencadenante de los procesos de desarrollo de los agregados y la funcionalidad biológica del suelo. La principal fuente de variabilidad que ha dirigido la organización de los procesos de desarrollo del suelo es la diferenciación espacial de la vegetación restaurada. No obstante, estos procesos también estuvieron condicionados por la presencia de pequeñas variaciones en la distribución de tamaños de las partículas minerales del suelo. Se han identificado reducciones de carácter exponencial asociadas a la intensidad de los procesos de erosión en diferentes parámetros analizados: AS, BR, DHA, PHA, β-GA y UA. Efectivamente, las restricciones impuestas por la erosión sobre el desarrollo vegetal dificultan considerablemente la incorporación de materia orgánica en el suelo; en consecuencia, los procesos de desarrollo y organización espacial de la estructura física y funcionalidad biológica del suelo se ven drásticamente limitados. Aunque los procesos de formación edáfica han estado activos en ausencia de procesos de erosión intensos, en las mejores situaciones analizadas los niveles de desarrollo fueron considerablemente bajos, probablemente debido a las pequeñas cantidades de materia orgánica acumuladas en el suelo (en general inferiores al 2%). Este resultado subraya la importancia de incluir enmiendas orgánicas dentro de las operaciones generales de restauración minera en ambientes mediterráneos.

Palabras clave: actividad enzimática, actividad microbiana, agregados del suelo, materia orgánica, restauración minera.

Development of soil physical structure and biological functionality in mining spoils affected by soil erosion in a Mediterranean-Continental environment

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Abstract

The main aim of this work was to analyse the development of soil functionality (aggregate development and soil biological functionality) on mining spoils 18 years after reclamation affected by different levels of soil erosion (rill erosion rate from 0 up to 70 t ha^{-1} $year^{-1}$) in a Mediterranean-Continental environment. For this purpose, different parameters were determined: the size distribution and stability of soil aggregates [mean weight diameter (MWD), microparticles and macroparticles proportions (MIP and MAP), and aggregate stability (AS)]; soil microbial population size [microbial biomass C (MBC)] and activity [basal respiration (BR) and dehydrogenase activity (DHA)]; and soil hydrolase activities [phosphatase (PHA), β-glucosidase (β-GA) and urease (UA)]. Soil ecological trends obtained from Principal Component Analysis revealed the accumulation of organic matter from restored vegetation as the triggering factor for the development of soil aggregation and biological functionality. The spatial differentiation of reclaimed vegetation was the principal source of variability for these soil forming processes, which were also conditioned by the presence of variations on soil particle size distribution. Erosion-related exponential decreases of AS, BR, DHA, PHA, β-GA and UA were identified. In fact, limitations of vegetation development caused by accelerated soil erosion drastically constrained the development and spatial organization of both physical structure and soil biological functionality, by preventing the accumulation of soil organic matter. Although soil forming processes in uneroded conditions occurred, soil functionality levels were rather low, probably due to the small amounts of soil organic matter reached, generally <2%. This stresses the importance of including organic amendments during restoration works in Mediterranean environments.

Key words: enzyme activity, microbial activity, mining restoration, organic matter, soil aggregates.

Introduction

Opencast mining is considered one of the most dramatic disturbances in terrestrial ecosystems (Nicolau and Moreno-de las Heras, 2005). A precondition for ecosystem reclamation in such highly disturbed mining areas is the development of functional soils with appropriate levels of organic matter and nitrogen, and active nutrient cycling (Bradshaw, 1997; Sourkova *et al*., 2005).

Both the activation of basic soil biological processes and the rearrangement of soil particles into stable aggregates are key factors related to the development of soil functionality. Soil microbes largely control the decomposition of soil organic matter and nutrient cycling (Filip, 2002). Similarly, soil aggregates control soil hydrology (Wu *et al*., 1990), affect soil oxygen diffusion and nutrient availability (Sextone *et al*., 1985), influence soil erodibility (Barthes and Roose, 2002) and constitute a pathway of organic carbon stabilisation and long term sequestration (Six *et al*., 2004).

Reclaimed land from opencast mining activities is particularly vulnerable to the effects of accelerated soil erosion processes (Haigh, 1992; Nicolau and Asensio, 2000). In fact, mining soils with high bulk density and massive structure, are generally characterised by low infiltration rates (Nicolau, 2002). This factor, and the presence of uncontrolled amounts of overland flow as a result of incorrect geomorphological designs, are frequently linked to the acceleration of soil erosion processes, particularly rill erosion (Evans *et al*., 1997; Moreno-de las Heras *et al*., 2008). The net loss of soil resources and drastic reduction of vegetation development are direct consequences of the unleashed degradation process in Mediterranean-Continental reclaimed mining systems (Moreno-de las Heras *et al*., 2005). Indeed, soil erosion has particularly destructive effects on soil forming processes (Pimentel *et al*., 1995).

Several parameters must be considered in order to determine the state and functionality of the soil system, since no individual parameter provides sufficient information (Gil-Sotres *et al.*, 2005). With respect to soil aggregation, the analysis of soil aggregate size distribution and stability can provide important information about soil quality (Sarah, 2005). Regarding soil biology, the quantity and activity of microorganisms are useful indicators of soil functions (Powlson *et al*., 1987). Additionally, information from a spectrum of hydrolytic enzyme activities provides valuable cues concerning element cycling that may influence vegetation development (García *et al*., 1994).

The objective of this work was to study the development of soil functionality (aggregate development and soil biological functionality) on mining spoils affected by different levels of soil erosion. For this purpose, soil physical parameters (reflecting soil aggregation), as well as several microbial and enzymatic parameters of the soil (reflecting the size and activity of microbial populations and the cycling of P, C and N elements) were analysed in five Mediterranean-Continental mining slopes (reclaimed 18 years ago) affected by a gradient of soil erosion. Vegetation composition and distribution were also considered, since differences in soil development processes have been frequently attributed to these factors (Cerdà, 1998; Puigdefábregas *et al*., 1999; Dornbush, 2007). The specific questions I wanted to answer were the following:

1- What are the triggering factors and sources of variability for soil functionality in mining spoils?

2- What is the impact of accelerated soil erosion processes on the development of functional soils?

Site description

This study was carried out in the Utrillas field site, which is located at the mine El Moral (Utrillas coalfield), central-eastern Spain (40º47'24'' N, 0º49'48'' W, 1100 m). The climate is Mediterranean-Continental. Mean annual air temperature is 11 ºC and the frost period runs from October to April. The moisture regime can be classified as Mediterranean-dry according to Papadakis (1966). Mean annual precipitation is 466 mm (concentrated in spring and autumn) and potential evapotranspiration is 758 mm.

The study site consists of five reclaimed mining slopes, all north-facing. These slopes were restored following the same procedures during 1988-89 by the Minas y Ferrocarril de Utrillas S.A. company. No topsoil replacement was carried out. Parent soil material consisted of a 100 cm layer of overburden mining spoil from the Utrillas cretacic formation, of Albian age (Gutiérrez, 1985). This is a clay-loam substrate (kaolinitic-illitic mineralogy) with basic pH (Table 8.1). Revegetation was undertaken by sowing a mixture of perennial grasses

Table 8.1. Basic characteristics (topography, soil, vegetation traits and erosion features) of the five experimental slopes. Values between brackets indicate S.D.

	N	Slope 1	Slope 2	Slope 3	Slope 4	Slope 5
Topography						
Slope gradient (º)		20	20	20	20	20
Aspect		North	North	North	North	North
Soil traits						
Stoniness (%)	25	22.2 (2.2)^{a}	24.7 (3.5)^{a}	26.2 (4.1)^{a}	25.2 (2.6)^{a}	24.5 (3.3)^{a}
Sand (%)	25	33.6 (3.6)^{a}	33.5 (3.7)^{a}	33.8 (3.0)^{a}	39.9 (1.8)^{a}	36.3 (2.7)^{a}
Silt (%)	25	26.9 (2.8)^{a}	33.8 (1.6)^{b}	30.8 (1.8)^{ab}	26.4 (2.9)^{a}	26.6 (4.5)^{ab}
Clay (%)	25	39.5 (2.2)^{a}	32.8 (2.9)^{b}	35.4 (2.1)^{ab}	33.8 (2.1)^{ab}	37.1 (2.9)^{ab}
Texture	25	Clay loam	Clay loam	Clay loam	Clay loam	Clay loam
Carbonates (%)	25	8.45 (1.32)^{a}	7.02 (0.54)^{a}	9.87 (2.26)^{a}	8.68 (1.49)^{a}	8.97 (1.23)^{a}
EC -w/v: ½- (dS m^{-1})	25	0.24 (0.13)^{a}	0.26 (0.14)^{a}	0.20 (0.10)^{a}	0.19 (0.03)^{a}	0.23 (0.03)^{a}
pH -H_2O; w/v: ½ -	25	8.03 (0.12)^{a}	7.96 (0.14)^{a}	7.95 (0.13)^{a}	7.95 (0.13)^{a}	7.91 (0.10)^{a}
Organic matter (%)	25	0.58 (0.20)^{a}	0.56 (0.23)^{a}	1.27 (0.35)^{ab}	1.46 (0.83)^{ab}	2.00 (0.74)^{b}
Total nitrogen (%)	25	0.04 (0.01)^{a}	0.03 (0.01)^{a}	0.07 (0.02)^{ab}	0.07 (0.04)^{ab}	0.10 (0.04)^{b}
C/N	25	8.88 (2.50)^{a}	10.24 (3.10)^{a}	11.38 (0.71)^{a}	11.51 (1.06)^{a}	12.05 (0.97)^{a}
Vegetation traits						
Cover [1] (%)	150	1.1 (2.0)^{a}	8.2 (5.5)^{a}	27.8 (9.9)^{b}	44.3 (16.2)^{bc}	59.4 (20.8)^{c}
Aerial biomass [2] (g m^{-2})	30	8.6 (12.2)^{a}	29.5 (20.5)^{ab}	65.3 (36.9)^{ab}	158.8 (60.8)^{b}	239.7 (95.3)^{b}
Litter [2] (g m^{-2})	30	0.3 (0.5)^{a}	9.8 (4.7)^{ab}	10.7 (17.9)^{ab}	23.8 (18.3)^{b}	34.9 (17.8)^{b}
Erosion features						
Rill density [3] (m m^{-2})	5	0.95	0.78	0.58	0.30	0.00
Rill erosion rate [4] (t $ha^{-1}year^{-1}$)	5	71.41	45.03	16.95	7.86	0.00

Abbreviations: N: number of samples; EC: electrical conductivity; w/v: relation weight (soil) / volume (water).
[1] Visually estimated in randomly distributed 0.25 m^2 plots during spring 2006.
[2] Oven-dried weight (60ºC, 72 h), collected in randomly distributed 0.25 m^2 plots during spring 2006.
[3] Linear rill length (m) per surface area (m^2).
[4] Measured from rill network dimensions following Morgan (1997).
Values with the same letters (a-c) within rows do not differ significantly at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney.

(*Festuca rubra*, *Festuca arundinacea*, *Poa pratensis* and *Lolium perenne*) and leguminous herbs (*Medicago sativa* and *Onobrychis viciifolia*). Soil preparation for sowing consisted of cross-slope ploughing without amendments (neither inorganic nor organic fertiliser).

Although these five slopes were restored using the same procedures, they currently differ in soil erosion (rill erosion rate from circa 0 to 70 t ha^{-1} year^{-1}) and rill network development (Table 8.1). These differences are mainly derived from different amounts of overland flow generated in water-contributing areas found at the top of each slope, as a result of an incorrect design. Therefore, there is a bare and almost flat area (slope angle: 4-6º, 6-9 m long) connected to the top of slopes 1 and 2. Similarly, a bare steep bank (slope angle: 40º, 3-7 m long) is connected to the top of slopes 3 and 4. No contributing areas were found at the top of slope 5.

Differences in past erosion rates between the slopes have produced notable differences in the development of reclaimed vegetation (cover, aerial biomass and litter) and in the accumulation of soil organic matter and total nitrogen (Table 8.1). Additionally, important differences were also found in the spatial distribution and identity of vegetative patches between the slopes (Table 8.2), due to erosion-related interference on vegetation succession (Merino-Martín, 2007; Moreno-de las Heras *et al*., 2008).

Methods

Soil sampling

Soil sampling took place in autumn 2006. In each experimental slope, five composite soil samples were collected from the top 10 cm, taking into account the proportional distribution of vegetative cover types within slopes (Table

Table 8.2. Description and spatial extent of vegetative cover types in the experimental slopes and associated distribution of composite soil samples for physico-chemical, structural, microbiological and biochemical analysis.

Vegetation type[1]	Cover (%)	Description	Aerial extent[2] (%)					Collected soil samples[3]				
			S1	S2	S3	S4	S5	S1	S2	S3	S4	S5
Bare soil (BS)	<15	Bare areas with some few scattered *Medicago sativa* individuals.	100	100	44	36	17	5	5	2	2	1
Sparse *S. chamaecyparissus* (SSc)	5-25	Areas sparsely covered by *Santolina chamaecyparissus* shrubs. Some annual grasses can occasionally appear, such as *Aegylops ovata*.	0	0	37	23	24	0	0	2	1	1
Dense *Graminae* (DG)	45-70	Sedimentation areas densely covered by perennial grasses, such as *Lolium perenne*, *Elymus hispidus* and *Dactylis glomerata*.	0	0	19	24	22	0	0	1	1	1
Dense *G. scorpius* (DGs)	45-70	Shrub patches densely covered by *Genista scorpius* and some associated perennial grasses, such as *Brachypodium retusum*	0	0	0	17	37	0	0	0	1	2

Slope codes: S1, slope 1; S2, slope 2; S3, slope 3; S4, slope 4; S5, slope 5.
[1] Defined following a detailed vegetation survey and multivariate analyses (Merino-Martín, 2007).
[2] Proportional aerial extent of each vegetative cover type per slope.
[3] Number of composite soil samples collected in each vegetative cover type per slope.

8.2). Each composite soil sample was formed by randomly collecting six subsamples along the slope from each selected vegetation type. To avoid alteration, samples were taken using a small shovel and stored in rigid plastic containers for transportation. Subsequently, soil samples were divided into two parts. One part was air dried to be used for physico-chemical characterization and soil aggregation analyses. The other part was sieved (2 mm sieve) and stored at 4ºC, and later used for microbial and biochemical assays.

Physico-chemical soil factors

All physico-chemical factors were analysed following the standardised methods proposed by the Spanish Ministry of Agriculture (MAPA, 1994). Thus, stoniness was determined as the proportional content of soil particles > 2 mm and textural particle size distributions were analysed following the Bouyoucos method. Soil electrical conductivity (EC) and pH were determined in a 1/2 (weight/volume) aqueous solution with a Crison® mod.524 conductivity-meter and a Crison® mod.2001 pH-meter, respectively. Total carbonates were measured using a Bernard calcimeter. Organic matter content and total nitrogen were determined by following the Walkley-Black and the Kjeldahl methods respectively.

Soil aggregation descriptors

The dry-sieving method proposed by Perfect and Blevins (1997) was used to characterize aggregate size distribution. About 300 g of air dried soil was shaken (60 s at 2 mm amplitude) in a vibrating sieve shaker (Endecotts® Octagon Digital) using a nest of sieves (0.25, 1.0, 2.0, 3.0, 4.0, 4.8 and 8.0 mm aperture). The percentage mass in each fraction was corrected for both rock fragment content and oven-dry (105ºC) moisture content. The mean weight diameter (MWD; Van Bavel, 1949) was calculated considering aggregate size distribution. The content of microparticles (MIP) and macroparticles (MAP) was calculated as the proportional fraction of particles <0.25 mm and >8 mm, respectively (Sarah, 2005). Finally, the stability of 4-4.8 mm size aggregates (AS) was determined using the water-drop test (Imeson and Vis, 1984). Following this method, the mean number of drops required to disrupt aggregates (20 air-dried aggregates from each soil sample, allowing them to pass through a 2.8 mm sieve) was determined and subsequently transformed to energy units using the equations proposed by Epema and Riezebos (1983).

Soil microbiological indicators

Microbial biomass carbon (MBC) was extracted from soil samples using the chloroform-extraction method (Gregorich *et al*., 1990). Extracted microbial carbon (C_{mic}) was measured using a total organic carbon autoanalyser (Shimadzu® TOC 5050A). Basal respiration (BR) was analysed by incubating 30g of soil moistened at 60% of its water holding capacity in hermetically sealed flasks (1 month at 28ºC). The CO_2 released was periodically quantified (daily during the first week and weekly thereafter) using an infrared CO_2 analyzer (Anagas CD 98 HR Plus; Environmental Instruments®). The cumulative amount of CO_2 released during the incubation period was calculated by adding all periodic measurements (Bastida *et al*., 2006). Dehydrogenase activity (DHA) was analysed following the method proposed by García *et al*. (1993), reducing 2-p-iodophenyl-3-p-nitophenil-5-phenyltetrazolium chloride in the dark to iodo-nitrotetrazolium formazan (INTF). INTF was measured in a spectrophotometer at 490 nm. Values were corrected for oven-dry (105ºC) soil moisture content, and used in statistical analyses.

Activity of soil hydrolases

Alkaline phosphomonoesterase activity (PHA) and β-glucosidase activity (β-GA) were determined following the methods proposed by Tabatabai and Bremner (1969) and Tabatabay (1982) respectively. Thus, p-nitrophenyl phosphate and p-nitrophenyl-β-D-glucopyranoside were used as substrate, quantifying spectrophotometrically at 400 nm the resulting p-nitrophenol (PNP). The buffered method proposed by Kandeler and Gerber (1988) was applied to measure urease activity (UA), using urea as substrate. The NH_4^+ formed was measured in a spectrophotometer at 690 nm after applying the modified Berthelot reaction. Values were corrected for oven-dry (105ºC) soil moisture content, and used in statistical analyses.

Statistical analysis

Principal component analysis (PCA; Jackson, 1993) was used to condense the variation comprised by soil aggregation parameters (AS, MWD, MAP, MIP), soil microbial indicators (MBC, BR, DHA) and the activity of soil hydrolases (PHA, β-GA, UA) in two orthogonal axes that summarised the main underlying gradients on soil data. The influence of physico-chemical soil characteristics on the two main gradients extracted by PCA was analysed by means of the Spearman´s R correlation coefficient. The most relevant factors were fitted to these gradients by means of second-order polynomial surfaces, using general additive models and interpolating fitted values onto the PCA biplot. In order to analyse the direct impact of soil physico-chemical factors on soil aggregation, and microbial and biochemical parameters, Spearman's R correlation coefficients were calculated. Finally, to assess the impact of accelerated soil erosion, differences between experimental slopes were tested using Kruskall-Wallis ANOVA and *post-hoc* Mann-Whitney U tests. The best fitting regression equation with the rill erosion rate was determined when significant differences between slopes were found.

STATISTICA 6.0 (Statsoft, 2001) was used for PCA and other statistical analysis. The ordisurf function of the Vegan package (Oksanen *et al*., 2007) was used to fit the second-order polynomial surfaces of relevant factors to the PCA biplot.

Results

Triggering factors and sources of variability

1- Analysis of soil gradients

The first two dimensions obtained from PCA explained 88.55% of the variance comprised in the data (component 1: 78.14% and component 2: 10.41%). The first PCA component was mainly determined by aggregate stability and the whole set of microbial and biochemical indicators (AS, MBC, BR, DHA, PHA, β-GA and UA contributed to 78% variability of component 1). Thus, this component reflected an ecological gradient of soil functionality, increasing microbial and hydrolytic enzyme activity, and soil aggregation quality towards its negative side (Fig. 8.1a). The second PCA component summarised the distribution of soil particles (MWD, MAP and MIP contributed to 76% variability of component 2), with mean weight

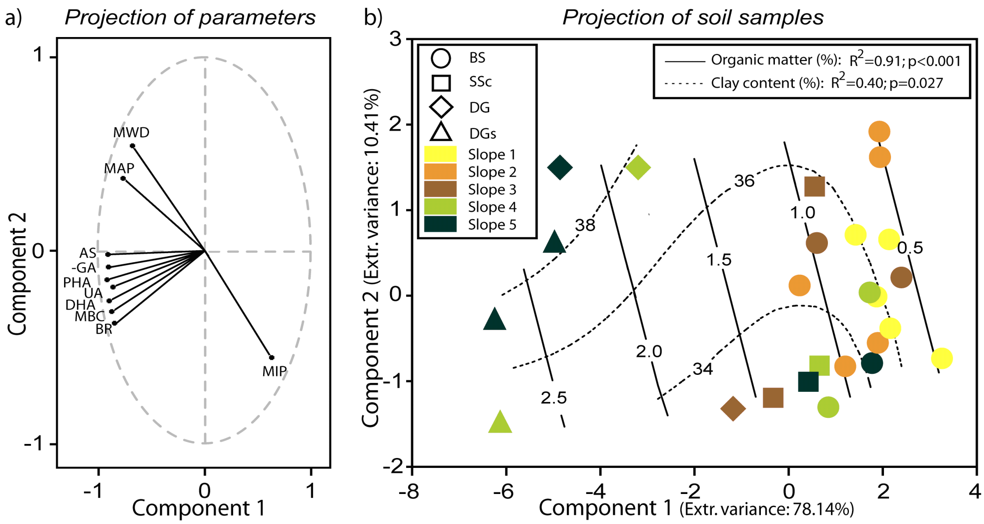

Figure 8.1. Projection of (a) soil parameters (structural, microbiological and biochemical) and (b) soil samples on the PCA biplot. Abbreviations of parameters (diagram a): AS, aggregate stability; MWD, mean weight diameter; MAP, macroparticles content; MIP, microparticles content; MBC, microbial biomass C; BR, basal respiration; DHA, dehydrogenase activity; PHA, phosphatase activity; β-GA, β-glucosidase activity; UA, urease activity. Vegetation types associated with collected soil samples (BS: bare soil, SSc: sparse *S. chamaecyparissus*; DG: dense *Graminae*; DGs: dense *G. scorpius*) are identified with different symbols (diagram b). Soil samples of the five experimental slopes are identified with different textures (diagram b). Soil organic matter and clay content (the two most relevant physico-chemical factors identified using Sperman´s R correlations) are represented by the contours derived from fitted second-order polynomial surfaces (diagram b).

diameter and macroparticles content opposite to soil microparticle content (Fig. 8.1a).

Soil samples were distributed within the PCA configuration according to slope and associated vegetation (specially along component 1). With regards to slopes, samples from the most eroded ones (slopes 1, 2 and 3) were located at the positive extreme of component 1, while samples from the most vegetated slopes (slopes 4 and 5) were distributed along the whole component (Fig. 8.1b). Regarding associated vegetation, the different types of soil cover were distributed following a sequence along component 1 (Fig. 8.1b). In this way, bare soils and soils barely covered by herbs or shrubs (BD and SSc types) were restricted to the positive half of component 1. On the other hand, soils densely covered by perennial grasses and shrubs (DG and DGs types) appeared in the negative half of component 1, associated with the highest levels of soil functionality. The main physico-chemical soil factors related to the ecological soil gradient expressed by this first component were those related to the organic soil component (organic matter and total N; Table 8.3). Likewise, a gradient of soil organic matter from values close to 0.5% to values around 2.5% (increasing towards the negative side of component 1) explained most of the variance summarised by PCA (R^2=0.91; Fig. 8.1b). No clear trends regarding slope and vegetation types were found along the second PCA component. Clay content was the main factor related to the physical gradient expressed by this component (Table 8.3). In this way, slight increases in clay content, from values lower than 34% to values around 38%, were

Table 8.3. Correlation coefficients (Spearman's R) between soil physico-chemical factors and the two first PCA components.

	Component 1	Component 2
Stoniness	-0.08	0.10
Sand	-0.48*	-0.32
Silt	0.33	0.01
Clay	0.13	**0.51****
Carbonates	-0.35	-0.13
Conductivity	-0.02	0.44*
pH	0.47*	-0.08
Organic matter	**-0.90*****	-0.31
Total N	**-0.89*****	-0.26
C/N	-0.37	-0.18

Coefficients in bold are >0.50.
*,**,***, significant at α=0.05; α=0.01 and α=0.001, respectively.

linked to the coarsening of soil aggregation reflected by PCA component 2 (Fig. 8.1b).

2- Impact of physico-chemical factors

Different physico-chemical soil factors were linked to physical soil aggregation variables (Table 8.4). MAP was positively correlated with organic matter content and total N, while these variables and clay content were negatively correlated to MIP. Likewise, AS was positively correlated with organic matter and total N. All microbial and enzymatic parameters were similarly linked to soil physico-chemical factors (Table 8.4). In this way, both organic matter and total N were strongly correlated with MBC, BR, DHA, PHA, β-GA and UA. Weaker positive correlations were found between all these parameters and sand content, and some negative correlations with silt content (particularly important in the case of UA) were found.

Impact of soil erosion processes

No significant differences were found between slopes with respect to the descriptors of aggregate size distribution (MWD, MAP, MIP; Table 8.5). Nevertheless, aggregate stability decreased significantly following an exponential decay trend along the soil erosion gradient comprised by the experimental slopes (Fig. 8.2a). Regarding soil microbiological and enzymatic indicators, no significant differences in the size of microbial populations (MBC) were found (Table 8.5). Conversely, the two microbial activity descriptors (BR and DHA), as well as the activity of the three analysed soil hydrolases (PHA, β-GA and UA) also showed

Table 8.4. Correlation coefficients (Spearman's R) between soil physico-chemical factors and variables related to soil aggregation, soil microbiology and the activity of soil hydrolases.

	Soil aggregation				*Soil microbiology*			*Soil hydrolases*		
	MWD	MAP	MIP	AS	MBC	BR	DHA	PHA	β-GA	UA
Stoniness	0.07	-0.12	-0.21	0.10	0.08	0.10	0.03	0.04	0.08	0.09
Sand	0.26	0.38	0.07	0.36	0.43*	**0.60****	**0.66*****	**0.62****	**0.62****	**0.54****
Silt	-0.22	-0.28	0.40*	-0.18	-0.27	-0.30	-0.44*	-0.42*	-0.37	**-0.52****
Clay	0.10	0.01	**-0.62****	-0.11	-0.11	-0.27	-0.19	-0.16	-0.23	-0.01
Carbonates	-0.06	0.06	-0.36	0.19	0.25	0.24	0.36	0.37	0.29	0.39
Conductivity	0.41*	0.31	-0.28	-0.03	-0.06	-0.02	-0.09	-0.05	-0.04	-0.11
pH	-0.41*	-0.45*	0.42*	-0.30	-0.43*	-0.46*	-0.39	-0.41*	-0.42*	-0.37
Org. matter	0.37	**0.55****	-0.45*	**0.67*****	**0.77*****	**0.89*****	**0.89*****	**0.92*****	**0.92*****	**0.93*****
Total N	0.39	**0.55****	**-0.54****	**0.67*****	**0.77*****	**0.85*****	**0.87*****	**0.92*****	**0.89*****	**0.94*****
C/N	0.11	0.21	0.02	0.24	0.32	0.44*	0.39	0.39	0.42*	0.33

Abbreviations: AS: aggregate stability; MWD: mean weight diameter; MAP. macroparticles content; MIP: microparticles content; MBC: microbial biomass C; BR: basal respiration; DHA: dehydrogenase activity; PHA: phosphatase activity; β-GA: β-glucosidase activity; UA: urease activity.
Coefficients in bold are >0.50.
*,**,***, significant at α=0.05; α=0.01 and α=0.001, respectively.

Table 8.5. Differences in structural and microbiological soil parameters, and the activity of soil hydrolases between the five experimental slopes. Values between brackets indicate S.D.

	N	Slope 1	Slope 2	Slope 3	Slope 4	Slope 5
Soil aggregation						
MWD (mm)	25	3.36 (0.39)[a]	4.03 (0.48)[a]	3.59 (0.47)[a]	4.06 (0.83)[a]	4.37 (1.06)[a]
MAP (%)	25	17.20 (6.68)[a]	24.58 (4.99)[a]	21.92 (5.63)[a]	30.97 (11.86)[a]	33.72 (15.35)[a]
MIP (%)	25	7.89 (1.26)[a]	8.93 (2.03)[a]	6.77 (1.32)[a]	8.12 (2.71)[a]	5.82 (3.00)[a]
AS (mJ)	25	8.15 (2.89)[a]	9.94 (1.94)[ab]	10.99 (4.62)[ab]	14.35 (7.37)[ab]	21.95 (10.81)[b]
Soil microbiology						
MBC (mg C_{mic} kg^{-1} soil)	25	104.51 (30.07)[a]	119.93 (34.04)[a]	141.41 (53.76)[a]	155.46 (62.24)[a]	172.60 (65.27)[a]
BR (mg CO_2-C kg^{-1} soil day^{-1})	25	2.87 (0.92)[a]	5.18 (3.10)[ab]	8.52 (5.31)[ab]	17.50 (16.55)[b]	19.45 (16.91)[b]
DHA (µg INTF g^{-1} soil h^{-1})	25	0.34 (0.12)[a]	0.36 (0.22)[ab]	0.56 (0.30)[ab]	0.87 (0.35)[ab]	0.98 (0.39)[b]
Soil hydrolases						
PHA (µmol PNP g^{-1} soil h^{-1})	25	0.65 (0.25)[a]	0.84 (0.63)[a]	1.55 (0.76)[ab]	2.82 (1.92)[ab]	3.79 (1.72)[b]
β-GA (µmol PNP g^{-1} soil h^{-1})	25	0.12 (0.05)[a]	0.23 (0.19)[ab]	0.44 (0.31)[ab]	1.03 (1.04)[b]	1.33 (0.80)[b]
UA (µmol N-NH_4^+ g^{-1} soil h^{-1})	25	0.21 (0.05)[a]	0.18 (0.13)[a]	0.58 (0.29)[ab]	0.68 (0.38)[ab]	0.92 (0.35)[b]

Abbreviations: AS: aggregate stability; MWD: mean weight diameter; MAP: macroparticles content; MIP: microparticles content; MBC: microbial biomass C; BR: basal respiration; DHA: dehydrogenase activity; PHA: phosphatase activity; β-GA: β-glucosidase activity; UA: urease activity.
Values with the same letters (a-b) within rows do not differ significantly at α=0.05. Tested using Kruskal-Wallis ANOVA and *post-hoc* Mann-Whitney.

significant exponential decreasing trends along the same erosion gradient (Fig. 8.2b-f). The variability of the analysed parameters also seemed to be affected by erosion processes, as standard deviations of all structural, microbial and enzymatic parameters were considerably low in the most eroded slopes (slopes 1, 2 and 3) compared with the least eroded ones (slopes 4 and 5; Table 8.5, Fig. 8.2).

Discussion

Triggering factors

Differences in the development of physical structure and biological functionality in the analysed mining soils mainly respond to differences in the amount of organic matter accumulated (Fig. 8.1a-b). Similarly, the accumulation of organic matter in mining spoils has been previously recognized as the key factor explaining the activation of soil biological processes (Gil-Sotres *et al*., 1995; Sourkova *et al*., 2005; Frouz *et al*., 2007). In this way, increases in soil organic matter and total N were associated with increases in microbial biomass carbon (Table 8.4), reflecting an increase in microbial population size (Powlson *et al*., 1987). Additional increases of basal respiration and dehydrogenase activity were associated with the organic soil component (Table 8.4), reflecting increases in actual microbial activity (García *et al*., 1997; Bastida *et al*., 2006). The development of these active microbial populations had a direct impact in the activation of vital processes for element cycling. Likewise, a direct association between the gradient of soil functionality triggered by organic matter (PCA component 1) and the enzymatic activity of phosphatase, β-glucosidase and urease was also found (Fig. 8.1a). Such hydrolytic enzymes, which are strongly dependent on the proliferation of microbial populations, are involved in the transformation of organic compounds, releasing inorganic forms of N and P directly available for growing plants (Burns, 1978).

Other factors related to soil texture can also influence both enzyme and microbial activity (Burns, 1978; Thomsen *et al*., 2003). In fact,

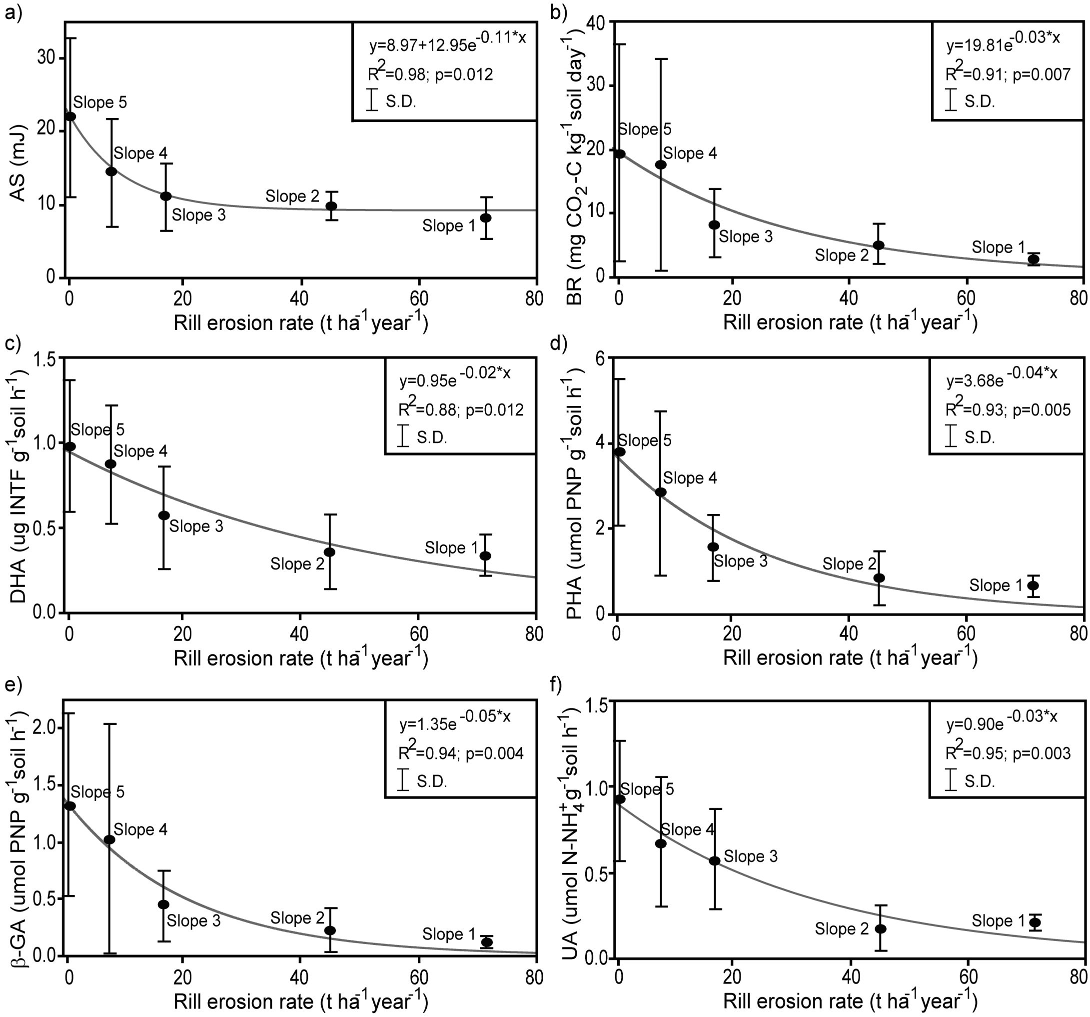

Figure 8.2. Best fitting regression equations between the soil erosion rate gradient represented by the five experimental slopes and the most contrasted structural, microbiological and biochemical soil parameters (identified testing the presence of significant differences between slopes with Kruskal-Wallis ANOVA). Abbreviations of parameters: AS, aggregate stability; BR, basal respiration; DHA, dehydrogenase activity; PHA, phosphatase activity; β-GA, β-glucosidase activity; UA, urease activity.

small variations in the proportion of silt and sand particles in clay and clay-loam textured soils could improve water circulation and soil aeration, enhancing biological soil processes (Porta *et al*., 1994). This could explain the positive correlations found between both microbial and enzymatic parameters and sand content, as well as the negative correlations between some of these parameters and silt content (Table 8.4).

The variability observed in soil aggregation (size distribution and stability) is mainly explained by variations in organic matter and clay content. This is not surprising, as clay and organic matter are the main soil bonding agents (Bronick and Lal, 2005). These bonding factors were associated with the coarsening of soil aggregates, by reducing the proportion of microparticles and increasing macroparticles (Table 8.4). Similarly, the straight link found between the stability of soil aggregates and soil organic matter (Table 8.4) can be explained by the cementing effect of the organic compounds and the stimulation of soil microorganisms (Cerdà, 1998; Six *et al*., 2004).

Although the impact of organic matter upon soil aggregation was significant, it was generally weaker than its impact upon microbiological and biochemical parameters (Table 8.4). This is probably because, soil microorganisms respond rapidly to changes in soil conditions by adjusting their structure, biomass and activity rates and conditioning the activity of hydrolytic enzymes (Schloter *et al*., 2003). On the other hand, the modification of soil physical traits (i.e. soil structure) is generally only detected when soil conditions undergo really drastic changes (Gil-Sotres *et al*., 2005). This fact points at the necessary integration of such soil microbiological and biochemical parameters with the classical structural-chemical ones for soil functionality assessment, especially when rapid soil ecological changes are analysed.

Sources of variability

In spite of the previously mentioned relationships between differences in soil texture, physico-chemical soil characteristics (excluding the organic component of the soil) were rather homogeneous between soil samples (Table 8.1). Consequently, other factors must be involved in the differentiation of soil forming processes. Such differences have been generally attributed to differences in vegetation structure, since plants determine the amount of soil organic matter by means of litter fall and root exudation (Sourkova *et al*., 2005; Dornbush, 2007). In the analysed case, the vegetation associated with the different types of soil cover was the main cause of variability in the ecological soil gradient described by PCA component 1, determining the differences in soil organic mater accumulation (Fig. 8.1b). In fact, the contribution to soil organic matter by growing vegetation has been crucial for the activation of soil forming processes in the studied mining slopes, where no organic amendments were carried out. Similar links between soil development and the spatial distribution of vegetation are frequent in water restricted environments, where the spatial variability of soil resources (water, organic matter and nutrients) and soil forming processes are intimately linked to vegetation patterns and structure (Puigdefábregas *et al*., 1999).

Impact of accelerated soil erosion

All processes involved in the development of soil physical structure and biological functionality were greatly affected by soil erosion. Likewise, exponential decreases in aggregate stability, microbial activity and enzymatic hydrolytic activities were found along the soil erosion gradient represented by the five experimental slopes (Fig. 8.2). These decreasing trends are in accordance with the differences found in soil organic matter and vegetation development between the slopes (Table 8.1). In fact, restored vegetation, which is the only source of soil organic matter in these artificial slopes, is affected by accelerated soil erosion processes through a reduction in water availability (Moreno-de las Heras *et al*., 2005). In this way, previous studies have evaluated different mechanisms by which soil erosion increases water loss through runoff in these reclaimed slopes, reducing water availability for plant growth and causing an exponential decline in biomass production: the reduction of water infiltration by surface crust formation and soil surface roughness reduction, and the efficient evacuation of runoff from the slopes by rill networks (Nicolau, 2002; Moreno-de las Heras *et al*., 2007). The final outcome of such erosion-related primary productivity limitations is the restriction of organic matter inputs into the soil (Pimentel *et al*., 1995). Moreover, soil erosion may also have direct effects upon soil organic matter. Indeed, erosion involves the preferential removal of the light and fine soil fractions, depleting soil organic matter and the most fertile layer of the soil profile (Lal, 2005; Marqués *et al*., 2008). As a result of the insignificant accumulation of soil organic matter (which is basic for the activation of soil forming processes), soil development processes in the studied highly eroded slopes (slopes 1, 2 and 3; rill erosion rate> 16 t ha^{-1} $year^{-1}$) can be considered inappreciable about 20 years after reclamation.

The lack of spatial differentiation of vegetative cover types in such highly eroded conditions also limited the development of variability on soil forming processes. Indeed, homogenously bare surfaces with some scattered individuals of Medicago sativa (a legume introduced by revegetation operations) are dominant in these highly eroded slopes, where spontaneous colonization and vegetation differentiation by ecological succession is severely constrained (Moreno-de las Heras *et al*., 2008). Conversely, spontaneous colonization has been possible in the less eroded slopes (slopes 4 and 5; rill erosion rate< 8 t ha^{-1} $year^{-1}$) and, owing to a dynamic process of vegetation succession, soil cover is spatially structured by bare and vegetated patches with different composition and density (Merino-Martín, 2007), leading to varying levels in the development of physical structure and biological functionality (Table 8.5).

Soil forming processes were active in these barely eroded slopes (slopes 4 and 5), specially in areas densely vegetated (DG and DGs cover types). Nevertheless, development levels of soil aggregation and biological functionality were considerably low when compared with those reported in natural Mediterranean-dry areas (Cammeraat and Imeson, 1995; Bastida *et al*., 2006), as would be expected in mining soils developed from overburden spoils (Gil-Sotres *et al*., 1995; Leirós *et al*., 1999). A reason behind this finding could be the small amounts of organic matter accumulated in the studied spoils. In fact, even in the case of soil samples associated with the most dense vegetation patches, levels of organic matter were only occasionally greater than 2%. This stresses the importance of including organic amendments during mining reclamation operations, specially when no topsoil replacement is carried out, as rates of soil organic matter accumulation driven by growing vegetation are particularly small in Mediterranean areas (García and Hernandez, 1997).

Conclusions

The accumulation of soil organic matter was the main trigger for the development of soil physical structure and biological functionality in the studied mining spoils, conditioning both the stabilisation of soil aggregates and the activation of the basic microbiological and element cycling processes. Nevertheless, other factors related to soil texture variations also affected these processes. In fact, clay content revealed an important physical influence, stabilising and coarsening soil aggregates, while both sand

and silt variations conditioned soil microbial and enzimatic activities, probably influencing water circulation and soil aeration. Due to the high sensitivity of the analysed microbial and biochemical parameters and the multiple soil ecological relations linked, integration of such soil biological factors with classical chemical-structural ones provided a high potential for soil functionality assessment.

The development and spatial differentiation of reclaimed vegetation played an essential role organising these soil development processes, as growing vegetation determined the incorporation of organic matter. Primary productivity decline and the spatial simplification of vegetative cover caused by soil erosion processes resulted in severe constraints for the development and spatial organisation of soil physical structure and biological functionality. Due to its dramatic consequences on soil forming processes, soil erosion control must be prioritised in reclamation planning for mining areas. Nevertheless, even in uneroded conditions development levels reached by soil aggregation and biological functionally were considerably low, also stressing the importance of including organic amendments during mining reclamation operations.

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Capítulo 9

Capítulo 9

Síntesis y conclusiones

Síntesis general

El propósito de esta sección es integrar los principales resultados obtenidos en los diferentes trabajos (**capítulos 3 a 8**) llevados a cabo para el desarrollo de los objetivos de la tesis. Éstos han dado lugar al análisis profundo de los efectos causados por el desarrollo de procesos acelerados de erosión hídrica superficial sobre los ecosistemas restaurados mineros de ambiente mediterráneo-continental, permitiendo incluso establecer relaciones cuantitativas acerca de la repercusión que la erosión tiene sobre parámetros ecosistémicos fundamentales. La integración de todos estos resultados permitirá cerrar la discusión abierta mediante el análisis de la presencia de umbrales críticos de carácter práctico (en términos de cubierta vegetal y tasas de erosión) que faciliten la predicción sobre la evolución de los ambientes restaurados de ladera que han sido objeto de estudio.

Escala regional (cuenca lignitífera de Teruel): Fuerzas directoras de la dinámica vegetal en laderas restauradas mineras. Papel de los procesos de erosión hídrica superficial.

Diferentes autores han señalado la falta de convergencia que en general caracteriza a la dinámica vegetal de los ambientes restaurados mineros, dada la multiplicidad de factores (determinísticos y/o de carácter contingente) que pueden actuar condicionando los procesos de sucesión vegetal (Wiegleb y Felinks, 2001; Novak y Prach, 2003; Nicolau y Moreno-de las Heras, 2005). En el caso de las laderas restauradas mineras de "la cuenca lignitífera de Teruel", los resultados obtenidos en el **capítulo 3** de esta memoria ponen de manifiesto un panorama complejo de evolución temporal a escala regional, caracterizado por el desarrollo de múltiples trayectorias sucesionales.

En grandes líneas se han identificado tres trayectorias principales de sucesión vegetal determinadas por las condiciones iniciales y el contexto ambiental de las laderas (ver Figura 3.3):

- En laderas con substratos ácidos, el establecimiento y desarrollo de la vegetación se ha visto limitado drásticamente, bloqueando la sucesión ecológica en fases muy iniciales (comunidad CT1; ver Tabla 3.2). Este efecto puede estar causado por la falta de especies adecuadas en las fuentes de propágalos cercanas (los suelos del entorno donde se desarrollan las comunidades vegetales naturales son en general de carácter básico) y, en algunos casos puntuales, toxicidad de los substratos (asociada a la presencia de materiales piríticos). Este problema ha sido detectado en minas abandonadas que operaron antes de la entrada en vigor del RD 2994/1982.

- Donde las condiciones ambientales son menos restrictivas (menor continentalidad climática y nivel de conservación elevado de la vegetación natural del entorno) y se empleó tierra vegetal de alta calidad, se han observado transiciones desde comunidades iniciales dominadas por vegetación herbácea (comunidad CT3; ver Tabla 3.2) a comunidades más diversas con diferentes especies de matorral (comunidades CT4 y CT8; ver Tabla 3.2), cuya composición y riqueza depende de la distancia a que se encuentran las fuentes de propágalos del entorno natural.

- Donde las condiciones ambientales son más restrictivas (mayor continentalidad climática y entorno natural degradado) y se realizaron operaciones de revegetación (con mezclas de semillas de especies herbáceas de rápido crecimiento), se ha observado que la incidencia de perturbaciones locales y los procesos de erosión son factores clave para la configuración de las trayectorias sucesionales. Así, las laderas no sometidas a procesos de erosión intensos tienden a desarrollar comunidades de herbáceas dominadas por las especies introducidas en las siembras iniciales (comunidad CT6; ver Tabla 3.2). En estas situaciones, fenómenos contingentes imprevistos, como el pastoreo y el efecto causado por enfermedades fúngicas sobre las poblaciones de especies dominantes (i.e.: *Medicago sativa*), han posibilitado la transición de estas comunidades vegetales hacia estados más diversos, que incorporan especies de matorral (comunidad CT7; ver Tabla 3.2). Por otro lado, en las laderas sometidas a intensos procesos de erosión en regueros se desarrollan comunidades vegetales muy ralas y poco diversas (comunidad CT5; ver Tabla 3.2). Estas formaciones vegetales están básicamente constituidas por unos pocos individuos de la especie leguminosa perenne *Medicago sativa* (introducida con las operaciones iniciales de revegetación), cuyo escaso desarrollo se limita a las áreas de inter-regueros de estas laderas erosionadas.

Los procesos de erosión en regueros en el marco regional analizado constituyen por tanto un fenómeno fundamental para la comprensión de las trayectorias de degradación en las comunidades vegetales restauradas. En este sentido, los efectos de la erosión sobre la vegetación constituyen un fenómeno característico de laderas mineras restauradas (Haigh, 1992; Nicolau y Asensio, 2000; Kapolka y Dollhopf, 2001). En el área de estudio, estos procesos de erosión intensos se encuentran asociados a laderas o taludes mineros con áreas de contribución de escorrentía en sus cabeceras o conectadas directamente con flujos de escorrentía procedentes de estructuras mineras externas (canales de guarda rotos, pistas mineras, plataformas y bermas superiores, etc.). Cabe señalar la importancia que este tipo de errores de diseño tienen para el desarrollo de procesos acelerados de erosión con formación de redes de regueros en laderas restauradas mineras (Porta *et al*., 1989; Evans, 1997; Hancock y Willgoose, 2004).

La identificación de la erosión hídrica superficial como una de las fuerzas directrices de la dinámica de la vegetación en estos ambientes restaurados dio pie a que fuera estudiada en detalle en los siguientes capítulos de la tesis.

Escala local (Área experimental Utrillas): Impactos causados por los fenómenos de erosión hídrica superficial con formación de redes de regueros sobre los ecosistemas restaurados.

Los trabajos desarrollados en el "Área experimental Utrillas" (**capítulos 4 a 8**) han puesto de manifiesto la importancia clave que los procesos intensos de erosión con formación de redes de regueros tienen sobre la organización de los ecosistemas restaurados a través de la reducción de los recursos hídricos disponibles para el desarrollo de la vegetación. En efecto, el desencadenamiento de procesos de erosión en regueros de diferente intensidad (asociados a la presencia de áreas de contribución de escorrentía de diferente tamaño en la cabecera de las cinco laderas estudiadas) ha condicionado, tras 15-20 años de evolución, la dinámica espacial de generación y circulación de los flujos de escorrentía superficial y sedimentos en estas laderas artificiales (**capítulos 4 y 5**), controlando la distribución y la disponibilidad de agua en el suelo (**capítulo 6**) y en consecuencia la dinámica y el desarrollo de la vegetación (**capítulos 6 y 7**), así como el desarrollo de las condiciones físicas y biológicas del suelo (**capítulo 8**).

El conjunto de las relaciones eco-hidrológicas analizadas en estas cinco laderas experimentales puede verse de forma condensada en la Figura 9.1. A continuación, en tres secciones consecutivas, se caracteriza la secuencia de impactos generados por los fenómenos de erosión en regueros sobre el desarrollo de estos ecosistemas restaurados de ladera.

1. Circulación de los flujos de escorrentía y sedimentos, y distribución asociada de la humedad en el suelo

De acuerdo con el modelo conceptual propuesto por Nicolau y Asensio (2000), entre las cinco laderas analizadas se pueden diferenciar dos situaciones extremas, cuya evolución temporal ha estado condicionada bien por el dominio de factores bióticos de control de los procesos hidrológicos (ladera 5), o bien por el dominio de factores de control abióticos de estos procesos (laderas 1 y 2).

En el caso extremo representado por la ladera 5, los niveles elevados de desarrollo vegetal (cubierta aprox. 60%, distribuida de forma discontinua a lo largo de la ladera) posibilitan a pequeña escala la regulación biológica de las respuestas hidrológicas del suelo, caracterizadas en este caso por niveles de erodibilidad bajos, así como por una capacidad de infiltración elevada y considerablemente estable entre estaciones (ver If y Sc en Tabla 4.3). A escalas mayores juegan un papel importante los fenómenos de re-infiltración y sedimentación, ambos muy activos sobre los flujos laminares de escorrentía y sedimentos que aparecen en esta ladera, con independencia del tipo de precipitación (ver Figuras 5.3 y 5.4). Estos fenómenos dan lugar a una importante redistribución de recursos (agua y suelo) a lo largo de la ladera, limitando de forma apreciable la salida de de los mismos fuera de las laderas. Así, las pérdidas de estos recursos representaron en las parcelas de mayor superficie estudiadas (parcelas de 3 x 15-m) menos del 5% de la precipitación pluvial en forma de escorrentía y aproxi-

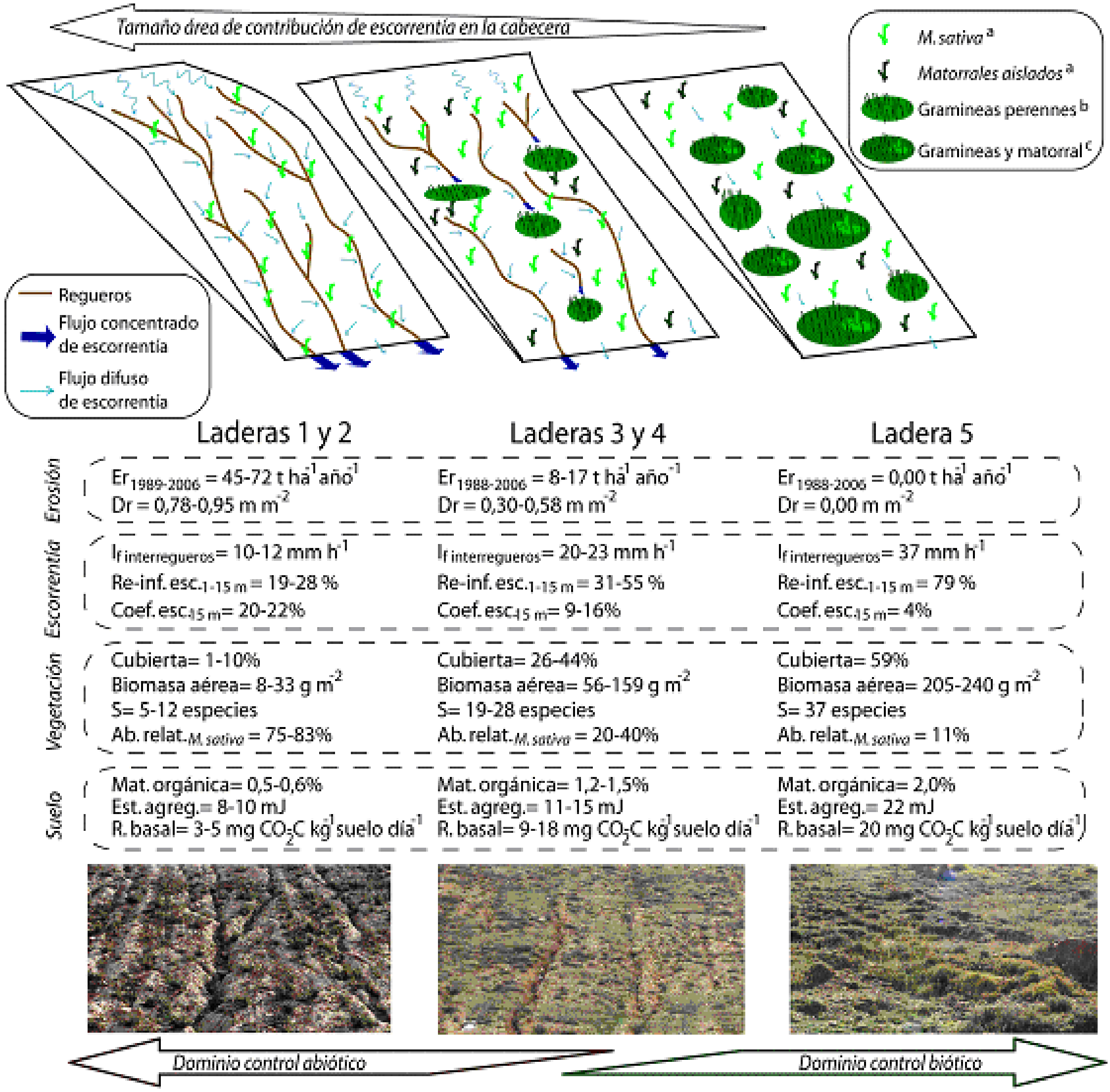

Figura 9.1. Diagrama que sintetiza de forma integrada el impacto de los procesos de erosión en regueros en el funcionamiento y organización de los sistemas artificiales de ladera analizados en el "Área experimental Utrillas". Abreviaturas para las variables caracterizadas: Er: tasas históricas de erosión en regueros; Dr: densidad de regueros; I_f: tasas de infiltración final del suelo (medias interestacionales año 2005-06) obtenidas con experimentos de lluvia simulada; Re-inf. esc.$_{1\text{-}15m}$: tasa de re-infiltración de los volúmenes acumulados anuales de escorrentía (2005-06) entre parcelas de 1 y 15 m de longitud; Coef. esc$_{15m}$: coeficiente de escorrentía acumulado anual (2005-06) obtenido en parcelas de 15 m de longitud; Ab. relat.$_{M.sativa}$: abundancia relativa (cubierta específica / cubierta total) de *M. sativa*; Estab. agreg.: estabilidad de los agregados (4-4,8 mm diámetro) del suelo; R. basal: respiración basal del suelo. Anotaciones en la leyenda descriptiva de la vegetación: a: individuos aislados de *Medicago sativa*; b: individuos aislados de *Santolina chamaecyparissus* y *Thymus vulgaris*; c: manchas de gramíneas perennes compuestas fundamentalmente por *Lolium perenne*, *Elymus hispidus* y *Dactilis glomerata*; d: manchas de vegetación mixta (gramíneas perennes y matorral) compuestas fundamentalmente por *Brachypodium retusum* y *Genista scorpius*.

madamente 2 t ha^{-1} de suelo en forma de sedimentos para el año hidrológico 2005-06 (ver Tabla 5.3). Estas pérdidas se encuentran dentro de los niveles habituales registrados en sistemas de ladera no degradados (tanto naturales como artificiales restaurados con éxito) de clima mediterráneo-seco (Puigdefábregas *et al*., 1999; Nicolau, 2002; Boix-Fayos *et al*. 2007; Martínez-Murillo y Ruiz-Sinoga, 2007).

En el extremo contrario, representado por las laderas 1 y 2 (tasas históricas de erosión en regueros: 45-72 t ha^{-1} año^{-1}), la circulación general de los flujos de escorrentía y sedimentos está condicionada a pequeña escala por la baja capacidad de infiltración del suelo y su alta erodibilidad, así como por la variabilidad estacional de estos parámetros (ver If y Sc en Tabla 4.3). En efecto, la falta de desarrollo vegetal limita la regulación hidrológica de estos suelos, los cuales presentan acusadas variaciones en la generación de escorrentía y sedimentos debido a las grandes fluctuaciones a las que las condiciones del suelo (estado de desarrollo de las costras y humedad) se ven sometidas en un ambiente mediterráneo-continental caracterizado por la acción invernal de los ciclos de hielo-deshielo y un desigual reparto estacional de las precipitaciones (ver Figura 4.2).

La presencia de redes de regueros espacialmente integradas y continuas condicionan a escalas mayores (de ladera) la circulación de los flujos de escorrentía y sedimentos en estas dos laderas intensamente erosionadas. Así, los flujos de escorrentía son capturados por estas redes, que aumentan drásticamente la conectividad hidrológica espacial, y son evacuados fuera de las laderas, reduciendo las posibilidades de re-infiltración de la escorrentía. Del mismo modo, la concentración de estos flujos aumenta la capacidad de arranque y transporte de partículas del suelo, incrementando el aporte de sedimentos a lo largo de la ladera a través de procesos activos de erosión en regueros. Estos fenómenos son especialmente nítidos en el caso de las precipitaciones de mayor intensidad (precipitaciones generadas por frentes atlánticos activos y tormentas convectivas; ver Figuras 5.3 y 5.4), que en conjunto representaron para el año hidrológico 2005-06 el 78% en volumen del total de precipitaciones. El resultado final de los condicionantes que imponen estas redes de regueros continuas es una pérdida neta de recursos en forma de escorrentía superficial y sedimentos. En este sentido cabe indicar que, a la escala mayor monitorizada (parcelas de 3 x 15-m), las pérdidas de agua y suelo para el año 2005-06 representaron en las laderas 1 y 2 más del 20% de la precipitación anual y más de 80 t ha^{-1} respectivamente (ver Tabla 5.3).

De acuerdo con los resultados obtenidos en otros sistemas altamente erosionados, el impacto hidrológico causado por estas redes de regueros tiene otra derivada más allá de maximizar la evacuación de agua y materiales de las laderas: influir de manera determinante sobre la distribución espacial de la humedad en el suelo (Nicolau, 2002; Biemelt *et al.*, 2005; Bracken y Crok, 2007). De este modo, se ha identificado un patrón emergente de distribución espacial de la humedad del suelo relacionado directamente con el patrón de geoformas: unidades de regueros e inter-regueros. Así, los contenidos de humedad registrados en los inter-regueros son muy reducidos y meramente superficiales en relación a los medidos en los regueros, por donde circulan concentrados los flujos de escorrentía (ver Figura 6.1). Estas diferencias espaciales de humedad son muy claras en el caso de las laderas con redes de regueros continuas (laderas 1 y 2). No obstante, estas diferencias tienden a desaparecer en las laderas en las que las redes de regueros están poco integradas espacialmente (laderas 3 y 4). La presencia de discontinuidades en los regueros (donde los flujos concentrados pasan a condiciones laminares) reduce la salida de recursos en forma de escorrentía y sedimentos de estas laderas y, al mismo tiempo, es la causa de la difuminación de este patrón característico de distribución espacial de la humedad.

2. <u>Respuesta de la vegetación al desarrollo de los procesos de erosión con formación de regueros</u>

La suma de los condicionantes que el desarrollo de los procesos de erosión en regueros imponen en la circulación de los flujos de escorrentía y distribución espacial de la humedad del suelo produce, como consecuencia directa, la reducción de la disponibilidad de agua para las plantas. En este sentido, se ha observado una fuerte reducción de carácter no lineal de la humedad disponible en los inter-regueros, asociada a la intensidad de los procesos de erosión analizados (ver Figura 6.2). Estas áreas (los inter-regueros) son los únicos ambientes donde es viable el desarrollo de la vegetación, dado que en los regueros el efecto erosivo de los flujos concentrados imposibilita el establecimiento y desarrollo de las plantas. El resultado inmediato es el aumento de los niveles de estrés hídrico en la vegetación de las laderas erosionadas (ver Figura 6.3 a y c).

El seguimiento de las fases de desarrollo de las plantas realizado en tres de las cinco laderas experimentales durante el ciclo de crecimiento vegetal 2003-04 apunta a esta reducción en la disponibilidad de agua como mecanismo clave para explicar los impactos causados por la erosión hídrica superficial sobre la vegetación. Así, considerando situaciones contrastadas en el gradiente de laderas estudiadas (la ladera 2 frente a la ladera 5) se pueden apreciar diferencias no sólo en la densidad de semillas en el suelo (78-350 semillas m^{-2} en la ladera 2 frente a aprox. 2000 semillas m^{-2} en la ladera 5; ver Tabla 7.2), sino también en la proporción de semillas del suelo que llegaron a desarrollar con éxito plantas adultas (2-13% en la ladera 2 frente a más del 80% en la ladera 5; ver Tabla 7.2) y en la producción de semillas que cierra el ciclo vegetal (considerando tanto la densidad como el tamaño de las semillas producidas; ver Tabla 7.3). La marcada disparidad identificada entre estas dos situaciones extremas en la proporción de semillas del suelo que llegan a desarrollar plantas adultas es consecuencia de las diferencias acumuladas sobre las tasas de emergencia y mortalidad de plántulas, vinculadas en ambos casos a las diferencias de disponibilidad de agua en el suelo. En este sentido cabe resaltar el papel preponderante que jugó la sequía frente a otras causas de mortalidad (congelación, enterramientos y arranques producidos por la erosión, y pisoteo del ganado) especialmente en los inter-regueros de las laderas más erosionadas, donde se llegaron a alcanzar tasas de mortalidad superiores al 70% (ver Tabla 7.2). Estos resultados se acercan a otros obtenidos en cárcavas naturales de ambiente mediterráneo-seco, donde las tasas de mortalidad de plántulas llegan a alcanzar valores del 98%, explicadas fundamentalmente por la escasez de agua en el suelo (García-Fayos *et al*., 2000).

Esta cadena de condicionantes (aportes de escorrentía superficial desde las cabeceras de laderas erróneamente diseñadas, que favorecen el desarrollo de redes de regueros, que disminuyen la disponibilidad de agua en el suelo para las plantas y la redistribuyen en detrimento de los inter-regueros) tiene consecuencias directas sobre la estructura de la vegetación desarrollada en las laderas estudiadas. No es azaroso que la composición florística de las laderas más erosionadas (laderas 1 y 2), donde la disponibilidad de agua es menor, esté restringida básicamente a una especie, la leguminosa perenne *M. sativa*, que representa más del 75% de la limitada cubierta vegetal de estas laderas (cubierta total 1-10%). Esta especie, que fue introducida mediante las operaciones iniciales de revegetación, presenta una especial capacidad para resistir periodos de sequía muy intensos, basada en un sistema radicular profundo y la capacidad de desprenderse de hojas y tallos para permanecer en estado aletargado durante periodos de estrés hídrico severo (Peake *et al*., 1975; Carter y Sheaffer, 1983; Bell *et al*., 2007). La formación de este tipo de comunidades vegetales muy simplificadas, constituidas por un reducido número de especies especialmente resistentes a la sequía, parece ser un resultado común a los procesos de degradación de la vegetación desarrollados en ambientes mediterráneos intensamente erosionados (Guardia, 1995; Guerrero-Campo y Montserrat-Martí, 2004).

No obstante, el reclutamiento de nuevos individuos de *M. sativa* en las laderas erosionadas estudiadas parece estar muy restringido a consecuencia del tiempo limitado y baja frecuencia con la que aparecen condiciones óptimas para la germinación de sus semillas (en términos de humedad del suelo y temperatura ambiental). La falta de reclutamiento de nuevos individuos, así como la falta de colonización por parte de otras especies (drásticamente limitadas como se ha indicado en las fases de emergencia y posterior supervivencia de plántulas) se encuentran detrás de la organización de estas comunidades simplificadas, espacialmente dependientes de los patrones de distribución de la humedad del suelo y perturbación mecánica de estas laderas. Efectivamente, los individuos supervivientes de *M. sativa* que componen la cubierta de vegetación de estas laderas intensamente erosionadas (laderas 1 y 2), aparecen mayoritariamente distribuidos en los bordes de los inter-regueros (ver representación gráfica en Figura 9.1). En estas posiciones las plantas maximizan sus posibilidades de supervivencia, ya que soportan niveles de estrés hídrico menores (posiblemente gracias al acceso a los mayores contenidos de humedad presentes en el entorno de los regueros cercanos; ver Figura 6.3 b y d), a la vez que evitan el impacto mecánico producido por los flujos concentrados de escorrentía que discurren por los regueros.

Este tipo de ajuste espacial en la distribución de la vegetación desaparece en las laderas en las que las redes de regueros son discontinuas (laderas 3 y 4). En éstas aparecen algunas manchas densas de gramíneas perennes (*Lolium perenne*, *Elymus hispidus* y *Dactilis glomerata*, en general procedentes de las siembras iniciales), desarrolladas en las discontinuidades de los regueros, donde se desparraman los flujos concentrados de escorrentía que circulan por los mismos (ver representación gráfica en Figura 9.1). En la ladera sin desarrollo de regueros (ladera 5), la composición y organización de la vegetación es más compleja (ver representación gráfica en Figura 9.1). Ésta se completa con la presencia de algunas especies de matorral dispersas en el espacio (*Santolina chamaecyparissus* y *Thymus vulgaris*), que con menor frecuencia aparecían ya en las dos laderas anteriores, así como otras manchas densas de carácter mixto compuestas por gramíneas perennes y matorral (*Brachipodium retusum* y *Genista scorpius*). Los procesos de erosión han jugado con certeza un papel muy determinante en la organización de la vegetación de estas laderas, regulando las posibilidades de desarrollo de las diferentes especies a través del control sobre la disponibilidad de agua; si bien es cierto que el paso esporádico de ganado por estas laderas puede haber tenido también una importancia relevante, especialmente explicando la llegada de semillas de algunas especies de matorral y gramíneas (Merino-Martín, 2007).

Se puede encontrar un importante número de evidencias en la literatura científica que, en concordancia con los resultados obtenidos en esta tesis, apuntan a la reducción de la disponibilidad de agua para las plantas es uno de los mecanismos fundamentales por los que la erosión hídrica superficial interfiere en el desarrollo de la vegetación (Thornes, 1985; Pimentel *et al*., 1995; Lal, 1998; Wilcox *et al*., 2003). El análisis del efecto de la intensidad de la erosión en regueros sobre los atributos principales de la estructura de la vegetación de los sistemas restaurados estudiados ha permitido establecer relaciones de carácter no lineal (exponenciales negativas) para la biomasa y diversidad de la vegetación (ver figura 6.6), de forma equivalente a las determinadas para la disponibilidad de agua en el suelo (ver figura 6.2). Las relaciones exponenciales negativas entre la erosión y el desarrollo de la vegetación ya habían sido planteadas teóricamente como las más cercanas a la realidad de los ambientes de clima mediterráneo (Thornes, 1990; Thornes y Brandt, 1994). Recientemente, éstas han sido incorporadas en las ecuaciones de los modelos dinámicos de competencia entre erosión y vegetación (Asensio y Nicolau, en revisión). Esta tesis las ha formalizado de forma empírica para el sistema artificial estudiado.

3. Efectos sobre el desarrollo de las funciones ecológicas del suelo

Diferentes trabajos han señalado la importancia central que la acumulación de materia orgánica tiene para la activación de las funciones ecológicas en los suelos mineros (Gil-Sotres *et al*., 1995; Sourkova *et al*., 2005; Frouz *et al*., 2007). En el caso estudiado, se puede afirmar que la incorporación de materia orgánica en el suelo ha actuado como factor precursor de la activación de la funcionalidad biológica (actividad microbiana y bioquímica que regula el reciclado de los elementos) y mejora de las condiciones físicas (estabilidad de los agregados) del suelo. En las cinco laderas analizadas, donde no se llevó a cabo ningún tipo de enmienda orgánica para su restauración, el desarrollo de la vegetación ha jugado un papel clave en los procesos de mejora de las condiciones biológicas y físicas del suelo, ya que representa la única fuente posible de materia orgánica.

Las limitaciones que han impuesto los procesos de erosión sobre el desarrollo de la vegetación en estas laderas tienen un efecto inmediato y paralelo sobre el desarrollo de la funcionalidad biológica y mejora de la estructura física de sus suelos. En analogía con la reducción de carácter exponencial previamente identificada para la biomasa de la vegetación, se han registrado caídas también de carácter exponencial en la estabilidad de los agregados del suelo y diferentes parámetros descriptores de la actividad microbiana del suelo (respiración basal y actividad de la deshidrogenasa) y reciclado de los elementos (actividad de diferentes hidrolasas: β-glucosidasa para el C, fosfatasa para el P y ureasa para el N) a lo largo del gradiente de erosión representado por las cinco laderas analizadas (ver Figura 8.2).

No obstante, incluso en las mejores condiciones (las representadas por la ladera 5) y tras casi 20 años de evolución, los niveles registrados de desarrollo de la estructura física y funcionalidad biológica del suelo son considerablemente bajos en comparación con los niveles comúnmente registrados en suelos naturales de clima mediterráneo seco (Cammeraat e Imeson, 1995; Bastida *et al*., 2006). Sin duda, la pequeña cantidad de materia orgánica que tras todos estos años ha sido capaz de incorporar en el suelo la vegetación (en los mejores casos, tan solo ligeramente superior al 2%), es la causa de estos bajos niveles de desarrollo. Esta situación es, por otro lado, la esperable en suelos artificiales desarrollados a partir de estériles mineros sin aplicación de enmiendas orgánicas para su restauración (Leirós *et al*., 1999).

¿Se pueden identificar unos umbrales críticos en términos de cubierta vegetal y tasas de erosión que permitan discriminar trayectorias de evolución distintas en los sistemas restaurados de ladera que se han analizado?

La identificación de umbrales críticos en las relaciones erosión-vegetación presenta un interés práctico notable para el mantenimiento y gestión de los nuevos entornos mineros restaurados. En el caso representado por los sistemas de ladera estudiados en el "Área experimental Utrillas", estos niveles críticos deben situarse en valores de desarrollo vegetal y tasas de erosión cercanos a la situación intermedia representada por las laderas 3 y 4 (cubierta vegetal= 26-44%, tasa de erosión en regueros= 8-17 t ha^{-1} $año^{-1}$; Figura 9.1). Estos umbrales coinciden a modo general con los resultados obtenidos en el **capítulo 4** de esta memoria. En este capítulo, y a partir de los experimentos realizados con lluvia simulada, se han identificado estadísticamente dos tipos de respuestas hidrológicas diferentes en base al control ejercido por la vegetación: con niveles de cubierta vegetal superiores al 50% las respuestas hidrológicas (generación de escorrentía y sedimentos) son controladas por la vegetación de forma eficiente, mientras que por debajo de niveles de cubierta del 30% no se produce control biológico de estas respuestas (ver Figura 4.5).

Por tanto, se podría manifestar que en situaciones caracterizadas por niveles de desarrollo vegetal superiores al 50% de la cobertura del suelo y tasas de erosión en regueros inferiores a 5 t ha^{-1} $año^{-1}$, la dinámica del sistema conduciría a situaciones de elevado desarrollo vegetal (la representada por la ladera 5; Figura 9.1), donde la vegetación realiza un control muy eficiente de los recursos (agua y suelo). Asimismo, con niveles de cubierta vegetal inferiores al 30% y tasas de erosión en regueros superiores a 20 t ha^{-1} año^{-1} la dinámica del sistema conduciría a situaciones máximas de degradación (las representadas por las laderas 1 y 2; Figura 9.1), donde el control de los recursos no obedece a la vegetación sino al desarrollo de los procesos de erosión con formación de regueros. Las situaciones intermedias, con niveles de desarrollo vegetal y tasas de erosión dentro del rango umbral definido, estarían caracterizadas por la falta de prevalencia entre los mecanismos de control abióticos y bióticos de sus trayectorias (situación representada por las laderas 3 y 4). Atendiendo a las consideraciones teóricas desarrolladas por Thornes (2004), estos estados intermedios están marcados por la baja predecibilidad de sus trayectorias de evolución final. En efecto, Wainwright *et al*. (2002), bajo el análisis de la dinámica de laderas naturales de clima semiárido, indican que este tipo de situaciones intermedias (con presencia de redes de regueros espacialmente discontinuas y niveles moderados de desarrollo vegetal) resultan poco estables y su evolución final está ligada al impacto de factores no deterministas (sequías, ciclos de años húmedos, eventos de precipitación extremos, etc.).

Claro reflejo de la operatividad de estos valores umbral (cobertura vegetal entre 30-50% y tasas de erosión en regueros entre 5-20 t ha^{-1} año^{-1}) en el contexto ambiental mediterráneo-continental analizado ("la cuenca lignitífera de Teruel") son los niveles de desarrollo tanto de la vegetación como de los procesos de erosión en regueros alcanzados por los diferentes grupos de laderas restauradas determinados en el **capítulo 3**, donde se identificaron las principales trayectorias regionales de sucesión vegetal (ver Apéndice 3.B). De este modo, el grupo de laderas intensamente erosionadas, cuya vegetación ha evolucionado hacia comunidades vegetales simplificadas (básicamente compuestas por individuos aislados de *M. sativa*; comunidad tipo CT5), está caracterizado en promedio por niveles de cubierta (suma de "vegetation cover" y "litter cover" en Apéndice 3.B) inferiores al 30% y tasas de erosión en regueros superiores a 20 t ha^{-1} año^{-1}. En este caso, tan sólo una intervención humana profunda, diseñada para limitar la pérdida de recursos hídricos del sistema, permitiría una salida del estado irreversible de degradación en el que estas laderas se encuentran. Por otro lado, todos los grupos de laderas cuya evolución ha resultado exitosa en términos de desarrollo vegetal (comunidades tipo CT4, CT6, CT7 y CT8) presentan en promedio niveles de cubierta cercanos o superiores al 50% y tasas de erosión en regueros inferiores a 5 t ha^{-1} año^{-1}. En último lugar cabe señalar los niveles intermedios de desarrollo vegetal y erosión típicamente registrados por los grupos de laderas recién construidas (comunidades tipo CT2 y CT3). La inestabilidad asociada a esta situación hace recomendable que, al menos durante los primeros 5 años de evolución, se realicen inspecciones periódicas sobre el desarrollo de la vegetación y formación de redes de regueros. De esta forma, se podría identificar de forma temprana la posible derivación de sus trayectorias de evolución hacia procesos sostenidos de degradación, reduciendo así los costes de intervención.

Diferentes trabajos, realizados dentro del marco climático mediterráneo en sistemas artificiales de ladera de características similares, señalan valores umbral semejantes a los obtenidos

(Andrés y Jorba, 2000; Loch, 2000). Estos valores críticos representan, por tanto, un criterio útil para el manejo y evaluación de las restauraciones aplicadas en ambientes mediterráneos. No obstante, es necesario considerar algunas limitaciones para la transferencia y empleo de los mismos. En este sentido, se debe recordar que estos umbrales prácticos de cubierta vegetal y tasas de erosión en regueros se han obtenido en sistemas artificiales de ladera (rango de pendiente aprox. 20-30º) de clima mediterráneo, restaurados en general con sustratos de texturas medias y niveles de desarrollo edáfico bajos, cuya cubierta vegetal está principalmente compuesta por especies herbáceas. Variaciones importantes respecto a estas condiciones pueden invalidar la utilidad de estos valores umbral. Efectivamente, numerosos trabajos remarcan la dependencia de este tipo de valores críticos respecto a las condiciones particulares de los sistemas analizados (Snelder y Bryan, 1995; Weltz *et al*., 1998; García-Fayos, 2004).

Conclusiones

1. Se han identificado diversas trayectorias de evolución en los sistemas restaurados de ladera derivados de la minería del carbón a cielo abierto en "la cuenca lignitífera de Teruel". Entre ellas destacan trayectorias hacia la degradación causadas por restricciones edáficas o por erosión hídrica superficial; trayectorias hacia comunidades estables dominadas por especies sembradas; trayectorias hacia comunidades diversas con especies de matorral y otras situaciones intermedias. Los principales factores de control implicados en la evolución de estos ecosistemas restaurados son: a) las condiciones de partida (errores de diseño, calidad de los substratos, tratamientos de revegetación aplicados); b) el contexto ambiental (continentalidad climática y nivel de conservación de los ecosistemas del entorno); c) otros factores de carácter eventual o contingente (efecto de plagas, pastoreo).

2. Los procesos de erosión hídrica superficial con formación de redes de regueros constituyen una fuerza directriz de las trayectorias de degradación en la evolución de estos ecosistemas restaurados. La presencia de áreas de contribución de escorrentía en la cabecera de las laderas, así como la ruptura de las estructuras de drenaje y/o conexión de las laderas con flujos de escorrentía externos, constituyen errores de diseño críticos para el desencadenamiento temprano de procesos intensos de erosión en regueros en estos sistemas artificiales.

3. El principal mecanismo por el que estos procesos de erosión actúan sobre la dinámica de los ecosistemas restaurados es la reducción de la disponibilidad de agua para las plantas, factor limitante en los ambientes mediterráneo-continentales. Efectivamente, el desarrollo de las redes de regueros incrementa la conectividad espacial de los flujos de escorrentía, maximizando la evacuación de la escorrentía superficial y con esto las pérdida de agua de las laderas. Al mismo tiempo, estas redes condicionan la distribución espacial de la humedad del suelo, concentrando los recursos hídricos en el entorno de los regueros. Como resultado final, se produce una fuerte reducción de carácter no lineal en la cantidad de agua disponible en los inter-regueros, únicas áreas donde el desarrollo de la vegetación es viable.

4. Como consecuencia directa de esta reducción en la disponibilidad de agua se produce un incremento de los niveles de estrés hídrico soportados por la vegetación y el bloqueo de los procesos de colonización vegetal. A escala de ladera, la suma de los efectos causados por la erosión en regueros provoca reducciones de carácter exponencial tanto en la biomasa de la vegetación como en su riqueza específica, asociadas en ambos casos a la disminución no lineal de la disponibilidad de agua para las plantas.

5. Las restricciones impuestas a la producción vegetal por el desarrollo de los procesos de erosión en regueros limitan a su vez la incorporación de materia orgánica en suelo, factor desencadenante del desarrollo de la funcionalidad biológica y mejora de la estructura física del suelo. Así, el impacto de los procesos de erosión sobre el desarrollo de las funciones ecológicas del suelo se ve reflejado también mediante caídas de carácter exponencial. En este

caso, estas caídas afectan tanto a los niveles de desarrollo de la estructura física del suelo (estabilidad de los agregados) como a diferentes parámetros microbiológicos y bioquímicos que regulan los procesos elementales de reciclado de los elementos en el suelo.

6. Resultado de la expresión no lineal de las relaciones erosión-vegetación en los sistemas restaurados de ladera que han sido objeto de estudio es la identificación de unos umbrales críticos precisos (en términos de cubierta vegetal y desarrollo de los procesos de erosión) con notable interés práctico para la identificación de trayectorias de evolución distintas:

- Con niveles de desarrollo de la cubierta vegetal inferiores al 30% y tasas de erosión en regueros superiores a 20 t ha^{-1} $año^{-1}$ se produce la transición hacia la formación de comunidades vegetales muy simples, ralas y poco productivas. En concreto, estas comunidades están dominadas por la especie *Medicago sativa*, una leguminosa perenne introducida mediante las operaciones de revegetación que presenta una elevada capacidad para soportar déficits hídricos intensos. Los individuos supervivientes de esta especie que integran la escasa cubierta vegetal de las laderas erosionadas tienden a distribuirse espacialmente en los bordes de los inter-regueros, posición que les permite minimizar simultáneamente los niveles soportados de estrés hídrico y perturbación mecánica. En estas situaciones los procesos de colonización natural se encuentran bloqueados por la baja disponibilidad de agua. Su recuperación, por tanto, precisaría de una intervención humana profunda enfocada a limitar las pérdidas de recursos hídricos en forma de escorrentía superficial.

- Con niveles de desarrollo de la cubierta vegetal por encima del 50% y tasas de erosión en regueros inferiores a 5 t ha^{-1} $año^{-1}$ se obtiene a escala de ladera un control biológico óptimo de los recursos hídricos. En estas situaciones la dinámica del sistema no se encuentra condicionada por los procesos de erosión hídrica superficial y su evolución temporal responde a otros procesos de tipo fundamentalmente biótico (interacciones entre especies, llegada de propágalos, perturbaciones locales, etc.).

Recomendaciones para el diseño y el manejo de las áreas restauradas

La importancia clave que las condiciones iniciales tienen en el éxito de las restauraciones mineras obliga a plantear un cuidadoso estudio y preparación previa tanto del diseño geomorfológico a aplicar como de los materiales y tratamientos que serán utilizados.

La aplicación de medidas para el control de los procesos erosivos, que inciden de forma determinante en la evolución de las áreas restauradas, debe partir desde el propio diseño geomorfológico de las laderas restauradas. El diseño de éstas no ha sido abordado dentro de los objetivos planteados en esta tesis; no obstante, los resultados obtenidos y diferentes aportaciones previas permiten señalar una serie de recomendaciones a tener en cuenta. Las diferentes medidas de diseño deben ir enfocadas a limitar la generación y circulación superficial de volúmenes importantes de escorrentía en las laderas, así como reducir la energía de estos flujos:

- La primera medida a tomar pasa por evitar la concentración de volúmenes importantes de escorrentía en la cabecera de las laderas (Hancock y Willgoose, 2004). Para esto se debe evitar la presencia de cualquier estructura que pueda actuar como área de contribución de escorrentía (plataformas desnudas, rampas de excesiva pendiente, pistas mineras, etc.) sobre las laderas restauradas. En caso de ser inevitable su presencia, los volúmenes generados por éstas deben ser derivados (mediante diseños en contra pendiente y/o canales de seguridad) a una red de regulación de los flujos (constituida por cunetas de recepción, cauces de transferencia y balsas de regulación) integrada en el conjunto de los espacios restaurados.

- En segundo lugar, se hace necesaria la aplicación de modelos de erosión para el diseño topográfico de las laderas restauradas, con el objeto de determinar los niveles generales de pendiente, perfil y longitud de ladera que en función de las condiciones climáticas locales, substratos y tratamientos empleados mantengan a los procesos de erosión en niveles aceptables, no limitantes para el desarrollo de la vegetación (Nicolau y Asensio, 2000). Los resultados obtenidos en esta tesis nos ofrecen unos niveles de erosión orientativos para dicho diseño en el contexto regional: a modo general se debe intentar ajustar el diseño de las laderas para obtener tasas de erosión en regueros inferiores o cercanas a 5 t ha^{-1} $año^{-1}$, y en ningún caso superiores a 20 t ha^{-1} $año^{-1}$. Entre los diferentes modelos disponibles para el diseño, destacan la versión adaptada de la Ecuación Universal de Pérdida de Suelo Revisada (RUSLE 1.06; Toy *et al*., 1999) y el modelo de deriva geomorfológica SIBERIA (Willgoose y Riley, 1998), los cuales han mostrado una elevada versatilidad para su aplicación en sistemas restaurados mineros (Nicolau 2003). Algunas consideraciones adicionales se pueden extraer de resultados obtenidos en diferentes trabajos donde se han aplicado estas herramientas de diseño: respecto a la forma de las laderas, parece existir un consenso amplio en la identificación de los perfiles cóncavos como la mejor alternativa de diseño para el control de la erosión (Hancock *et al*., 2003); asimismo, bajo las condiciones climáticas mediterráneo-continentales propias de la región donde se ha llevado a cabo esta tesis y los substratos en general disponibles para su restauración, se desaconseja el diseño de laderas con pendientes superiores a los 20º (MFU, 2001).

Los resultados obtenidos en esta tesis a escala regional indican que el empleo en superficie de materiales con restricciones químicas severas compromete el éxito de las restauraciones. Efectivamente, la selección y empleo de buenos materiales edáficos para cubrir las formas topográficas creadas constituye un nuevo punto crítico para el diseño de estos ecosistemas artificiales de ladera. La mejor alternativa es el extendido de espesores de "tierra vegetal", ya que estos substratos poseen en general condiciones edáficas bien desarrolladas y una importante reserva de propágulos vegetales. No obstante, la disponibilidad de "tierra vegetal" suele ser muy limitada, por lo que con frecuencia es necesario recurrir al empleo de estériles de mina. En caso de recurrir a esta alternativa es necesario realizar una cuidadosa selección previa tanto del material como del lugar de aplicación. Así, todo material estéril cuyas características físico-químicas puedan representar una limitación seria para el desarrollo de la vegetación (acidez, texturas extremas, salinidad, presencia de metales pesados u otro tipo de elementos tóxicos, etc.) debe ser rechazado para su empleo como material de superficie. Del mismo modo, como regla general se debe evitar el empleo de este tipo de materiales sin formación edáfica en las condiciones más restrictivas (solanas, laderas de elevada pendiente, etc.). Una medida importante a tener en cuenta para acelerar los procesos de mejora de las condiciones físicas y activación microbiana de estos substratos es la incorporación, mediante enmiendas, de materia orgánica en el suelo, de cuya presencia depende el desencadenamiento de los procesos de desarrollo de la funcionalidad edáfica en estos materiales estériles.

La siembra de mezclas de semillas de especies herbáceas de rápido crecimiento es una medida ampliamente utilizada en los espacios restaurados mineros, con el objeto de favorecer el desarrollo temprano de la cubierta vegetal y controlar así el desarrollo de los procesos de erosión (Loch, 2000; Holl, 2002). Esta medida resulta especialmente necesaria en caso de aplicar estériles de mina como substrato edáfico en la superficie de las laderas artificiales, ya que estos materiales no contienen semillas ni ningún otro tipo de propágulo vegetal. Los resultados obtenidos indican que el empleo de mezclas comerciales compuestas por especies alóctonas de rápido crecimiento inhiben el progreso sucesional de la vegetación, limitando la incorporación de otras especies, en especial los matorrales. Como medidas generales a tener en cuenta para el diseño de estas siembras se puede indicar la necesidad de reducir en las mezclas la proporción de semillas de aquellas especies que muestren una mayor persistencia (como es el caso de *Medicago sativa*) y la inclusión de especies de carácter autóctono, entre éstas algunas especies de matorral que hayan mostrado una capacidad de colonización exitosa (i.e.: *Genista scorpius*, *Dorycnium pentaphyllum*, *Santonina chamaecyparissus*). Otra alternativa sería la producción de perturbaciones de carácter local (pequeñas rozas, introducción de pastoreo a baja intensidad, etc.) en las comunidades dominadas por las especies sembradas que faciliten la entrada de las especies presentes en el entorno.

En ausencia de los condicionantes impuestos por los procesos de erosión sobre el desarrollo vegetal así como del bloqueo sucesional producido por las siembras iniciales, el progreso de las comunidades vegetales desarrolladas en estos sistemas artificiales de ladera puede verse limitado también por la llegada de propágalos de nuevas especies. En este caso tienen una gran importancia tanto el estado de conservación de las comunidades del entorno inalterado como la distancia a éstas. De este modo, la aplicación de medidas de conservación en la vegetación circundante a los espacios afectados, así como la inclusión de manchas remanentes de vegetación inalterada dentro de los entornos mineros, podrían facilitar, mediante la disminución de las distancias existentes desde las áreas fuente hasta las áreas restauradas, la incorporación de nuevas especies y el progreso sucesional de las comunidades vegetales restauradas.

El manejo de estos espacios restaurados debe tener en cuenta también otros fenómenos y factores de carácter contingente. Un caso importante entre éstos es la incorporación en las áreas restauradas de nuevas actividades y usos no previstos, como es el caso del pastoreo. Los resultados obtenidos parecen indicar que la introducción de esta actividad podría favorecer el progreso sucesional de las comunidades restauradas, actuando como vector de dispersión de especies presentes en áreas naturales lejanas y limitando la dominancia de las especies introducidas mediante las siembras iniciales. Este efecto positivo fue identificado en las cuencas mineras presentes en el municipio de Utrillas, bajo unas condiciones muy concretas. Los terrenos restaurados en aquel momento (años 2002 y 2003) estaban controlados directamente por la empresa minera MFU y el paso del ganado por las laderas restauradas se producía tan sólo de forma eventual. Posteriormente, estos terrenos restaurados fueron cedidos al municipio y la población local, quienes incrementaron la carga ganadera en las áreas restauradas. El estado actual de estas áreas dista mucho del que presentaban cinco años atrás: se ha producido una degradación intensa de la cubierta vegetal y el aceleramiento de los procesos de erosión hídrica superficial. Por tanto, la introducción de esta actividad en los espacios restaurados debe realizarse con mucha cautela, y en todo caso quedar supeditada al desarrollo de protocolos de investigación específicos que determinen bajo qué condiciones puede ser introducida.

Dada la multiplicidad de resultados y patrones de evolución que pueden desarrollarse en los espacios restaurados mineros se hace necesaria la inclusión de procedimientos de vigilancia y control tras la ejecución de las restauraciones, con el objeto de adaptar el manejo de estos ecosistemas artificiales a sus situaciones particulares (Bradshaw, 1988; Holl y Cairns, 2002). El uso de una serie de indicadores de estado fácilmente cuantificables permitiría la identificación temprana de situaciones de bloqueo y/o degradación que impidan un desarrollo óptimo de las áreas restauradas, facilitando así la toma de decisiones para su gestión. Los umbrales críticos de cubierta vegetal y tasas de erosión en regueros determinados en esta tesis pueden servir como criterio práctico para la valoración del estado de estos sistemas artificiales de ladera, así como de otros de características similares en el contexto climático mediterráneo seco. En este sentido cabe recordar que en situaciones con niveles de cubierta vegetal (suma de vegetación y hojarasca) inferiores al 30% e intensidades de erosión en regueros superiores a las 20 t ha^{-1} $año^{-1}$ los procesos de colonización y desarrollo vegetal se encuentran completamente bloqueados por la pérdida masiva de recursos hídricos en forma de escorrentía superficial. En estos casos debe realizarse una intervención humana profunda para reducir la pérdida de agua del sistema. En último lugar, se debe señalar que durante los primeros años de evolución se debe realizar una vigilancia especialmente frecuente e intensa de la situación de estos sistemas artificiales, debido a la elevada inestabilidad dinámica que en general caracteriza los estados iniciales de estas laderas y la baja predecibilidad de la evolución futura de las mismas.

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Una sola puerta de tres, abierta.
Una sola puerta.
Enfrente, la montaña.
Pasa la nube inmensa;
toda suya... todo suyo.
Huracanes de vientos;
lluvia andante semiparalela
y en todo el monte funerales alegres, naturales,
de hojas muertas.

Los cabellos terráqueos danzan todos iguales
al son de trompetas invisibles que vienen de los mares.

Llegó el otoño; llegó la muerte...
¡Mas no para todos!
Hoy morirán hojas y animales.

Mas no morirán para siempre y, en su transformación de mañana
darán
con más calor
a la tierra,
de su muerte,
pasado mañana,
brotes de esperanza.

¡Y yo no he muerto!
Me alegro de la lluvia
y me alegro del viento.
Si tengo frío, me caliento;
si tengo miedo, ¡Que no lo tengo!,
susurro y pienso...
y para mañana
ya me he comido mi pequeña ración de esperanza.

Una sola puerta de tres, abierta.
Una sola puerta inmensa.

Manuel Muñoz Sánchez