TESIS DOCTORAL

Efectos de las cenizas y la repelencia al agua en la hidrología de suelos afectados por incendios forestales en ecosistemas mediterráneos

Ash and water repellency effects on soil hydrology in fire-affected Mediterranean ecosystems

Programa de Doctorado

Problemas actuales del territorio valenciano y técnicas de análisis

Merche Berenguer Bodí

Departament de Geografia. Universitat de València
Directores: Artemi Cerdà, Jorge Mataix-Solera y Stefan H. Doerr
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Esta memoria ha sido presentada por Mercedes Berenguer Bodí, licenciada en Geografía, para aspirar al grado de Doctor.

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Esta tesis ha sido dirigida por los profesores abajo firmantes, el Dr. Artemi Cerdà del Departament de Geografia de Universitat de València, el Dr. Jorge Mataix-Solera del Departamento de Agroquímica y Medio Ambiente de la Universidad Miguel Hernández de Elche y el Dr. Stefan H. Doerr del Departamento de Geografía de la Swansea University (Reino Unido).

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Resumen

El **fuego** es un factor ecológico natural en la Tierra y ha actuado con diferente intensidad desde hace 400 millones de años. En condiciones de clima mediterráneo el fuego es un factor recurrente que las sociedades humanas han controlado desde hace milenios. Sin embargo, tras medio siglo de abandono agrícola y ganadero, la gestión del monte mediterráneo mediante el uso del fuego es inexistente. Además, con la recuperación de la vegetación, el fuego afecta ahora a una densa y continua masa forestal, convirtiendo los incendios en un riesgo por la dificultad en ser controlados y por su elevada intensidad. Este tipo de incendios descontrolados son los que pueden inducir efectos ambientales adversos al producir la alteración de las propiedades edáficas y favorecer elevadas tasas de erosión, potenciando procesos de Desertificación. Además, el fuego también pone en riesgo propiedades y vidas humanas. Esta Tesis Doctoral trata algunos de los aspectos ambientales de los **incendios forestales**, con especial atención a los **suelos**. Sin duda, el estudio de las aspectos biofísicos como los aquí presentados, junto con estudios socio-económicos, permitirán una mejor gestión de las zonas afectadas por incendios, y con ello la planificación del territorio.

Los cambios que produce el fuego en la vegetación, la micro y macrofauna, los ciclos biogeoquímicos, las propiedades del suelo y los procesos hidrológicos y geomorfológicos, se deben al calentamiento producido por el mismo incendio, pero también a las condiciones ambientales surgidas tras la pérdida de la cubierta vegetal y el recubrimiento con cenizas. Es decir, el paso del fuego transforma súbitamente el aspecto y el funcionamiento del ecosistema allí donde se produce, y deja una herencia que lo afectará durante años de manera muy dinámica y variable dependiendo de múltiples factores.

Las cenizas son uno de los factores clave para entender la evolución de las zonas afectadas por incendios forestales. En el contexto de este estudio, las **cenizas** se consideran el residuo del material orgánico tras el paso del fuego. Estas cubren el suelo temporalmente formando una capa de espesor y características físicas y químicas variables, las cuales dependen del material original y de la severidad del incendio. Las cenizas influyen decisivamente en los ciclos biogeoquímicos, incluido el del carbono, modifican las tasas de infiltración del suelo y la generación de escorrentía, controlan las tasas de erosión y alteran las propiedades físicas y químicas de los suelos. Incluso después de ser lavadas, disueltas o erosionadas, seguirán influyendo en los ciclos de la materia y la energía al modificar las propiedades del agua y de los suelos donde se incorporen. Sin embargo, son uno de los elementos más desconocidos del ecosistema post-incendio ya que fueron relativamente olvidadas por la investigación científica.

Contrariamente, se estudió la erosión y producción de escorrentías, los cambios en la materia orgánica y la agregación del suelo o la evolución de la vegetación de forma intensiva y reiterada, pero los estudios llevados a cabo sobre incendios forestales sólo habían considerado las cenizas como un factor relevante en el ciclo de nutrientes y como fertilizador del suelo. Muy pocos estudios habían explorado los **efectos de las cenizas** en la hidrología y erosión del suelo y, de la investigación que se está llevando a cabo, todavía no se han obtenido conclusiones definitivas. Lo que ahora sabemos sin duda es que las propiedades de las cenizas presentan una gran variabilidad, y con ello respuestas diversas e incluso contrastadas.

El **objetivo general** de esta Tesis Doctoral, "*Efectos de las cenizas y la repelencia al agua en la hidrología de suelos afectados por incendios forestales en ecosistemas mediterráneos*", es estudiar los efectos de las cenizas sobre los procesos hidrológicos, edáficos y geomorfológicos. Se ha prestando especial atención a la repelencia al agua, tanto de las propias cenizas como a sus efectos en la repelencia al agua del suelo al ser factores decisivos para entender su papel como gestores del agua, sedimentos y nutrientes. La tesis se presenta como un compendio de **cinco artículos científicos** realizados entre julio de 2008 y junio de 2012. Tres de estos trabajos han sido publicados en revistas internacionales de alto índice de impacto (Geoderma y Catena), y todos los resultados fueron presentados en congresos nacionales o internacionales para su discusión publica antes de su redacción.

El primer trabajo y punto de partida de la Tesis Doctoral, fue una revisión bibliográfica sobre el efecto del fuego en las propiedades del suelo publicada en el Boletín de la AGE (Asociación de Geógrafos de España) y titulada "*Efectos de los incendios forestales en la vegetación y el suelo en la cuenca mediterránea: revisión bibliográfica*". Tras poner en contexto esta investigación, se procedió a la realización de experimentos y estudios concretos. En el primero y cuyos resultados están en prensa en la revista Catena con el título de "*Spatial and temporal variations of water repellency and probability of its occurrence in calcareous Mediterranean rangeland soils affected by fires*", se estudió la evolución de la repelencia al agua en el suelo inmediatamente tras un incendio y en cuencas quemadas 10 y 20 años antes, en un ecosistema mediterráneo con suelo calcáreo. Se examinaron además las variaciones mensuales debidas a los cambios en la humedad del suelo y las variaciones espaciales debajo de diferentes tipos de vegetación y en microparcelas debajo de un mismo individuo y se comprobó que las cenizas afectan a la repelencia al agua tras el incendio. También se calcularon las probabilidades de encontrar repelencia al agua en estos suelos. Este experimento sugirió nuevas cuestiones sobre si las cenizas podían ser repelentes al agua y, como no existían estudios sobre ello, este fue el objetivo del siguiente trabajo "*The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content*" publicado en Geoderma. Mediante experimentos de laboratorio se estudió también la relación de la repelencia al agua de las cenizas con el contenido de carbono orgánico y su color, y cómo afecta la incorporación de cenizas a la repelencia al agua del suelo. Llegado este punto se creyó necesario además conocer su efecto en la repelencia al agua del suelo cuando las cenizas lo cubren. Por tanto, se procedió a la realización de varios experimentos más intentando aportar a la vez información sobre su papel en la generación de escorrentías, en la composición química de éstas, y en la erosión producida tras un incendio, cuando diferentes cenizas cubren diferentes suelos con capas de varios espesores. Los resultados obtenidos con los experimentos del laboratorio están en prensa en la revista Geoderma con el título "*Hydrological effects of a layer of vegetation ash on underlying wettable and water repellent soil*", mientras que los obtenidos con experimentos en el campo están publicados en las Actas del III Congreso Internacional FESP "*Fire effects on soil properties*" en Guimaraes (Portugal) con el título "*Runoff rates, water erosion and water quality from a soil covered with different types of ash*".

Esta Tesis Doctoral aporta al conocimiento científico una serie de contribuciones de entre las que destacan las siguientes:

1) En cuanto a las **variaciones espaciales y temporales de la repelencia al agua** en suelos calcáreos mediterráneos y la predicción de la probabilidad que esté presente en el suelo:

- La repelencia al agua puede ser una propiedad común en los suelos calcáreos de los bosques mediterráneos. En las zonas estudiadas, la presencia de la repelencia al agua del suelo osciló estacionalmente entre 40-95% con algunas muestras manifestando repelencia extrema, pero disminuyendo en profundidad. Asimismo, se encontraron incrementos en la repelencia del suelo a 1 cm de profundidad tras un incendio forestal, aunque la repelencia al agua se redujo a niveles muy bajos tras un año tanto en superficie como en profundidad. Este cambio pueden debido al lavado de sustancias hidrofóbicas o a su degradación, al recubrimiento de la superficie del suelo con sedimentos producto de la erosión post-incendio que están compuestos de partículas del suelo y cenizas hidrofílicas y a la falta de deposición de material orgánico hidrofóbico fresco al no haber temporalmente vegetación como consecuencia del incendio. Los resultados indican que el restablecimiento de la repelencia al agua a niveles de antes del incendio tarda más de 10 años según los resultados de esta investigación en la Sierra de Enguera (Valencia).

- La variabilidad de la repelencia al agua del suelo bajo *Pinus halepensis, Quercus coccifera* y *Rosmarinus officinalis* en la misma zona de estudio de 1 es igual de amplia que entre individuos de la misma especie y dentro de una parcela 10 cm × 10 cm. La menor variabilidad aparece en suelos desnudos donde la mayoría de las muestras fueron hidrofílicas. La repelencia al agua también es también variable según la humedad del suelo, siendo mayor en condiciones secas de verano.

- A partir de covarianzas y factores fijos simples de cuantificar (tipo de vegetación, profundidad del suelo y humedad del suelo) se desarrolló un modelo estadístico que permite estimar la probabilidad de encontrar repelencia al agua en el suelo en diferentes estaciones y condiciones meteorológicas en bosques mediterráneos maduros. El modelo obtenido puede representar una herramienta útil aunque se requiere más estudio y comprobaciones para poder aplicarlo a otras áreas.

2) En referencia a las **propiedades de las cenizas** estudiadas:

- Las cenizas procedentes de incendios forestales pueden ser repelentes al agua. La mayor presencia y persistencia de repelencia al agua en cenizas de bosques de *Pinus halepensis* y vegetación asociada aparece en las recogidas en incendios forestales de baja severidad y, de las producidas en el laboratorio en las generadas entre 200-300 °C, mientras que a más de 400°C, la repelencia es inexistente. Además, las cenizas producidas a partir de *Quercus coccifera* y *Pinus halepensis* resultan en mayores niveles de presencia y persistencia de repelencia al agua que las de *Rosmarinus officinalis*. La repelencia al agua en las cenizas se reduce tras su humedecimiento. Esta propiedad está relacionada con la proporción de carbono orgánico/inorgánico y el tipo de componentes orgánicos. No obstante, según nuestros resultados el color no es un buen indicador tanto de la repelencia al agua como de la cantidad de carbono orgánico si se comparan cenizas de diferentes incendios forestales.

- La cantidad de carbono orgánico total en las cenizas en este estudio oscila entre 4,6% y 31,1% y en las cenizas producidas en el laboratorio entre 22,9% y 60,3%. Los mayores valores de carbono orgánico en las cenizas generadas en el laboratorio pueden deberse a que el proceso de combustión producido no es directamente comparable al ocurrido en incendios forestales.

3) Respecto a la **longevidad de las cenizas en el suelo**, se ha comprobado que una capa de cenizas de más de 5 mm recogidas en un incendio de baja severidad, continúa cubriendo el suelo tras dos simulaciones de lluvia en el laboratorio con una intensidad de 82 mm h^{-1} durante 40 min y con una pendiente de 10º. Igualmente, grosores de más de 5 mm de cenizas producidas a elevadas temperaturas en el laboratorio mantuvieron el suelo cubierto tras dos lluvias simuladas en el campo de 55 mm h^{-1} durante 1 hora con una pendiente de 5º. Grosores de menos de 5 mm dejaron el suelo descubierto tras estos eventos.

4) En cuando al efecto de las cenizas en la **repelencia al agua del suelo**:

- Si las cenizas cubren el suelo, una capa de cenizas hidrofílica de más de 5 mm de grosor reduce considerablemente la repelencia al agua del suelo al incrementar la presión hidráulica y el contacto entre el agua y el suelo, y por tanto promueve flujos preferenciales a lo largo del perfil del suelo. Tras la primera lluvia y en condiciones de suelo húmedo, las cenizas no producen una reducción de la repelencia al agua del suelo significativa.

- Si las cenizas están incorporadas en el suelo, pueden aumentar o reducir la repelencia al agua del suelo dependiendo su naturaleza hidrofílica o hidrofóbica. En el caso de cenizas hidrofóbicas, la repelencia resultante puede ser elevada si se incorporan al suelo mediante mecanismos exentos de agua, como la acción eólica o la bioturbación.

5) Respecto a los efectos de una capa de **cenizas** en la **respuesta hidrológica**:

- Las cenizas pueden aumentar la arroyada superficial si la conductividad hidráulica ($K_{cenizas}$) es menor que la intensidad de la lluvia, lo que puede ocurrir debido a la compactación y encostramiento de las cenizas cuando tienen un alto contenido en carbonato cálcico o si las cenizas son repelentes al agua.

- En todos los otros casos, cuando $K_{cenizas} >$ intensidad de la lluvia, y sobre todo para la primera lluvia tras el incendio, las cenizas retrasan el inicio de la arroyada proporcionalmente al espesor de la capa debido a su elevada capacidad de almacenamiento de agua.

- Una vez las cenizas están saturadas, se produce un flujo superficial y subsuperficial entre las cenizas y el suelo hasta que ambas capas se equilibran y pasa a dominar el proceso la capacidad de infiltración del suelo.

- Sin embargo, las capa de cenizas puede modificar la capacidad de infiltración del suelo (a), incrementándola respecto a un suelo no cubierto de cenizas porque previenen el suelo de encostrarse y pueden reducir la repelencia al agua de éste (ver punto 4) o (b) reduciéndola si taponan los poros del suelo, según el tamaño de tamaño de partícula de ambos materiales.

6) Las **cenizas protegen el suelo** de la erosión producida por el impacto de las gotas de lluvia en todos los casos y previenen la erosión laminar siempre que no se saturen y no ocurra escorrentía superficial. Esta reducción es especialmente importante sobre suelos repelentes desnudos tras un incendio, que son los que suelen producir las mayores tasas de erosión. El sedimento producido en suelos cubiertos por cenizas está asociado con la escorrentía superficial generada (ver punto 5), pero durante episodios de lluvias intensas, las cenizas pueden aumentar la capacidad de transporte del sedimento contribuyendo a eventos extremos de erosión.

7) Los **nutrientes** liberados por las cenizas modifican la calidad del agua de escorrentía, incrementando el pH, la conductividad eléctrica y especialmente la cantidad de cationes. Los nutrientes solubilizados no son exactamente los mismos que los existentes en la composición original en forma de cenizas ya que dependen del volumen de la escorrentía producida (ver punto 5), de la solubilidad de los compuestos de las cenizas y de las interacciones químicas con el suelo. Con la primera lluvia, si esta es suficientemente abundante, la mayor parte de los nutrientes son solubilizados y lixiviados o arrastrados con la escorrentía, pero algunos de ellos pueden aumentar tras unas semanas al producirse cambios químicos tanto en las cenizas como en el suelo.

Abstract

Fire is a natural phenomenon that has affected the Earth's land surface to varying degree over the last 400 millions years. Under the climatic conditions of the Mediterranean, fire has been a recurrent component that human societies have controlled since millennia. Recently, however, following 50 years of rural depopulation, land management using fire has largely disappeared here and with forest regrowth, fire now affects a dense and often contiguous biomass. Wildfire in the Mediterranean has thus become a risk due to its high intensity and the difficulty in controlling it. It is particularly the intense, uncontrolled fires which can result in adverse environmental effects, modifying soil properties, promoting high erosion rates and hence desertification. In addition, fire endangers properties and human lives. This PhD thesis addresses some environmental aspects of **wildfire** with special consideration of its **effects on soils**. The study of the environmental aspect presented here together with social and economic studies will allow a better management of fire-affected areas and the regional planning.

Fire can result in changes in vegetation, micro and macro fauna, biochemical cycles, soil properties, and hydrological and geomorphological processes due to the heat input, but also due to the environmental conditions after the loss of vegetation and the added cover with ash. Wildfire produces a sudden transformation to the characteristics and functioning of the ecosystem, and leaves an influence that can affect it for years in a very dynamic and variable way depending on multiple factors.

Ash is one of the key elements affecting burned areas. In the context of this study, ash is the residue of the organic material after the fire and blankets the soil temporally with a layer of variable thickness, physical and chemical properties, which depend on fuel characteristics and fire severity. It is of major importance for the biogeochemical cycles, including the carbon cycle. It modifies soil infiltration and runoff generation, affects erosion rates, and changes physical and chemical soil properties. Even after being removed from the soil surface by slope wash, wind erosion or dissolution, ash may still affect the matter and energy cycles, and can modify the properties of soils if incorporated. Despite its implications, ash has not seen much attention in the scientific studies of wildfire and it remains one of the least understood elements of the post-fire ecosystem. In contrast, changes in soil organic matter and soil aggregation, runoff and erosion, and the recovery of vegetation, have been intensively and repeatedly studied. To date, ash has been considered largely only as an important element in the nutrient cycle and as a soil fertiliser. Very few studies have explored **ash effects** on soil hydrology and erosion, and from the research to date, few definitive conclusions have

been obtained. What we do know now is that ash properties are very variable, which results in different and sometimes contrasting effects.

The **general objective** of this PhD thesis, "*Ash and water repellency effects on soil hydrology in fire affected Mediterranean ecosystems*" is to study the effects of ash on pedological, hydrological and geomorphological processes. Particular attention is given to the wettability of ash and its effects on soil water repellency because of their potential relevance in controlling water, sediment and nutrient fluxes in the fire affected ecosystem. The thesis is presented in **five scientific papers** based on work carried out between July 2008 and June 2012. Three of them have been published in international journals (Geoderma and Catena), and all have been presented at national and international conferences for their public discussion before publication.

The starting point of the Thesis was a bibliographic review on the effect of fire in soil properties published in the "Boletín de la AGE" (Journal of the Spanish Geographers Association) with the title "*Efectos de los incendios forestales en la vegetación y el suelo en la cuenca mediterránea: revisión bibliográfica*". After putting the research in context, different studies and experiments were carried out. In the first one, which results are in press in Catena with the title: "*Spatial and temporal variations of water repellency and probability of its occurrence in calcareous Mediterranean rangeland soils affected by fires*", it was studied the changes in soil water repellency immediately after a forest fire and in catchments burned 10 and 20 years ago. This work also examined the seasonal variations in water repellency related with soil moisture and spatial variations under different vegetation types and within microplots under the same vegetation type, and proved that ash can affect soil water repellency after a fire. The probability of water repellency occurrence was also calculated. This work raised new questions about water repellency of ash, and since no studies had been published on this issue, this became the main objective of the following part of this work. Its outcomes are now published in Geoderma under title "*The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content*". Involving laboratory experiments, this work also examined the relationships between ash water repellency, organic carbon content and its colour, and how ash incorporation into the soil affects soil water repellency. At this point, the need became evident to also investigate the effects of ash on water repellency when ash covers the soil. Therefore, different experiments were carried out to explore, at the same time, the role of ash in runoff generation and chemical composition of runoff water and its role in the erosion produced after the fire when different types of ash cover different soils with various thickness. The results obtained from the related laboratory experiments are in press in the journal Geoderma, "*Hydrological effects of a layer of vegetation ash on underlying wettable and water repellent soil*", and the results obtained in the field experiments are published in the Proceedings of the III International FESP Conference "*Fire effects on soil properties*" in Guimaraes (Portugal) with the title "*Runoff rates, water erosion and water quality from a soil covered with different types of ash*".

This PhD Thesis contributes to the scientific knowledge with the following general findings:

1) Regarding the **spatial and temporal variations of water repellency** in Mediterranean calcareous soils and the prediction of its probability of occurrence:

- Water repellency can be a common soil property in the calcareous soils of Mediterranean forests. Soil water repellency in the study area ranged seasonally from 40-95 % spatial coverage. Some samples showed extreme water repellency, but this was generally reduced at 1 cm depth. An increase in water repellency was found at 1 cm depth after a forest fire, however, water repellency decreased to very low levels at the surface and at depth after one year. It is suggested that this decrease was due to the gradual washing out of hydrophobic substances and the lack of fresh organic and hydrophobic material inputs after fire, together with the topsoil's cover of wettable material composed of ash and soil. The results obtained at the Sierra de Enguera study sites indicate that the recovery to the pre-fire water-repellent conditions may take more than 10 years.

- Variability of soil water repellency under *Pinus halepensis, Quercus coccifera* and *Rosmarinus officinalis* within the same site of 1 ha was as high as within the same vegetation type and as within a plot of 10 cm × 10 cm. The lower variability was found in bare soil plots, which were mostly wettable. Water repellency was also variable depending on soil moisture, being highest under dry summer conditions.

- A statistical model that allows estimation of the probability of water repellency occurrence in mature Mediterranean calcareous soils was derived using covariates and fixed factors that are straightforward and economic to measure (vegetation type, soil depth and moisture content). This is a simple model that may represent a powerful tool to estimate water repellency occurrence under different seasonal and meteorological conditions. Further work is required to determine the wider applicability of this model to other areas.

2) Concerning **ash properties**:

- Ash from vegetation fires can be water repellent. A greater frequency and persistence of water repellency in ash from *Pinus halepensis* and associated vegetation forests was found for a low severity fires and, in laboratory generated ash the generated at temperatures between 200–300 °C, whereas above 400 °C repellency was absent. Ash generated from *Quercus coccifera and Pinus halepensis* litter exhibited higher water repellency levels compared to *Rosmarinus officinalis*. Ash water repellency might be reduced after wetting. Water repellency was related to the ratio of organic/inorganic carbon content in ash and type of organic carbon compounds. The colour of ash was found not to be a good predictor of either water repellency or total organic carbon content when ash from different wildfires was compared.

- The organic carbon content in ash from wildfire in this study ranged from 4.6% to 31.1% and from the ash produced in the laboratory from 22.9% to 60.3%. The higher contents in ash generated in the laboratory ash might be the result of a combustion process that is not directly comparable to that experienced in the field.

3) **Regarding the longevity of ash on the ground**, it was shown that an ash layer of more than 5 mm produced in a low severity wildfire remained on the ground following two laboratory rainfall simulations of 82 mm h^{-1} during 40 min with a slope of 10º. Similarly, thicknesses of more than 5 mm of ash produced in the laboratory at very high temperature remained covering the soil after two field rainfall simulations events of 55 mm h^{-1} with a slope of 5º. Ash layers with lower thicknesses left the soil uncovered after these events.

4) Regarding the effects of ash on **soil water repellency**:

- If ash covers the soil, a layer of wettable ash of more than 5 mm thickness can reduce soil water repellency by increasing the hydraulic pressure and the contact between water and soil, and hence promoting fingered subsurface flow especially during the first rain event after the fire. After the first rainfall event, and if the soil is still wet, ash did not produce a significant reduction on soil water repellency compared with bare soil.

- When ash is incorporated into the soil, it can increase or reduce soil water repellency depending on its wettable or water repellent nature. In the case of water repellent ash, the increase can be especially important if ash is incorporated by dry mechanisms as such as wind erosion and deposition, or bioturbation.

5) With respect to the effects of an ash layer covering the soil on the **hydrological response**:

- Ash may enhance overland flow if the hidraulic conductivity of ash (K_{ash}) is lower than rainfall intensity, which can happen when ash is compacted and crusted due to its high content of calcium carbonate, or when ash is water repellent.

- In all the other cases, when K_{ash}> rainfall intensity and especially for the first rainfall event, ash delays the onset of overland flow proportionally to its thickness due to its high water storage capacity.

- Once ash is saturated, overland flow and subsurface flow between the ash and soil starts until both layers reach an equilibrium, after which the soil infiltration rate dominates the process.

- Soil infiltration rates can be modified by ash, (a) resulting in a higher soil infiltration rate by decreasing the degree of soil water repellency (see point 4) or by reducing the soil's susceptibility to crusting (ash acts as a mulch), and (b) reducing the soil infiltration rate by enhancing pore clogging, which depends on the particle sizes of soil and ash.

6) Ash does **protect the soil** from splash erosion and prevents sheet erosion as long as the ash layer is not saturated and overland flow does not occur. This reduction is especially important for bare water repellent soils, for which erosion is usually higher than for wettable soil conditions after a fire. The sediment yield produced from soils covered with ash is associated with overland flow (see point 5). During intense rainfall events, and if ash is already saturated, ash can increase the transport capacity of the sediment, contributing to high erosion events.

7) The **nutrients** released from ash can modify water quality increasing its pH, electrical conductivity and especially cation content. The nutrients solubilised are not necessarily the same as the elemental composition of ash itself. Runoff composition depends on the volume of water produced (see point 5), on the solubility of the ash components and on the chemical interactions with water from rainfall and soil. After the first intense rain event, most of the nutrients are solubilised and lixiviated or washed out, however, some of them may increase in the runoff or soil water some weeks later due to chemical interactions with water from rainfall and soil nutrients.

Resum

El **foc** és un factor ecològic natural en la Terra i ha actuat amb diferent intensitat des de fa 400 milions d'anys. En condicions de clima mediterrani, el foc és un factor recurrent que les societats humanes han controlat des de fa mil·lennis. Tanmateix, després de mig segle d'abandonament agrícola i ramader, la gestió de la serra mitjançant l'ús del foc és inexistent i amb la recuperació de la vegetació, els incendis forestals afecten ara a una densa i continua massa forestal. El foc així es converteix en un risc, per la dificultat de ser controlat i per la seva elevada intensitat. Aquest tipus d'incendis descontrolats són els que poden induir efectes ambientals adversos ja que alteren les propietats edàfiques i afavoreixen elevades taxes d'erosió, potenciant doncs processos de Desertificació. A més, el foc també posa en risc propietats i vides humanes. Aquesta Tesi Doctoral tracta alguns dels aspectes ambientals dels **incendis forestals** i presta especial atenció als sòls. Sense cap dubte, l'estudi dels aspectes biofísics presentats ací juntament amb altres estudis socioeconòmics, permetran una millor gestió de les zones afectades pels incendis, i alhora una més encertada planificació del territori.

Els canvis que produeix el foc en la vegetació, la micro i macrofauna, els cicles biogeoquímics, les propietats del sòl i els processos hidrològics i geomorfològics, són deguts a l'escalfament produït pel propi incendi, però també a les condicions ambientals produïdes per la pèrdua de la coberta vegetal i el recobriment de les cendres. És a dir, el pas del foc transforma sobtadament l'aspecte i el funcionament de l'ecosistema afectat i deixa una herència que l'afectarà durant anys de manera molt dinàmica i variable segons multitud de factors.

Les cendres són un dels factors clau per a entendre la evolució de les zones afectades per incendis forestals. En aquest estudi les **cendres** es consideren el residu del material orgànic cremat. Estes cobreixen el sòl temporalment formant una capa d'espessor i característiques físiques i químiques variables, les quals depenen bàsicament del material original i de la severitat de l'incendi. Les cendres influeixen decisivament en els cicles biogeoquímics, incloent el del carboni, modifiquen les taxes de infiltració i la generació d'escolament, controlen les taxes d'erosió i alteren les propietats físiques i químiques del sòl. Fins i tot, després de ser llavades, dissoltes o erosionades, seguiran influint en els cicles de la energia i la matèria ja que modificaran les propietats de l'aigua o el sòl on s'incorporen. Malgrat això, les cendres són un dels elements més desconeguts del ecosistema post-incendi ja que foren relativament oblidades per la investigació científica. Contràriament, en les zones afectades per incendis forestals es van estudiar de forma intensiva i reiterada l'erosió i producció d'escolament, els canvis en la matèria orgànica i l'agregació del sòl i l'evolució de la vegetació, però els estudis

realitzats fins ara sols havien considerat les cendres com a un factor rellevant en el cicle dels nutrients i com a fertilitzant. Molts pocs estudis havien explorat els **efectes de les cendres** en la hidrologia i erosió del sòl. De la investigació que s'està realitzant ara al respecte, encara no s'han obtingut conclusions definitives, però ja sabem sense cap dubte que les seves propietats presenten una elevada variabilitat i per tant poden produir una diversitat d'efectes, alguns fins i tot contrastats.

L'objectiu general d'aquesta Tesi Doctoral, "*Efectes de les cendresi la repel·lènciaa l'aigua en la hidrologia de sòls afectats per incendis forestals en ecosistemes mediterranis*", és l'estudi dels efectes de les cendres sobre els processos hidrològics, edàfics i geomorfològics. S'ha prestat atenció especial a la repel·lència a l'aigua de les cendres i als seus efectes en la repel·lència a l'aigua del sòl ja que són factors decisius per a comprendre el seu paper com a gestors de l'aigua, sediments i nutrients. La Tesi es presenta com a un compendi de **cinc articles científics** realitzats entre juliol 2008 i juny de 2012. Tres d'aquests treballs s'han publicat en revistes internacionals (Geoderma i Catena) i tots els resultats han sigut presentats prèviament en conferències nacionals i internacionals per a la seva discussió pública.

El primer treball i punt de partida de la Tesi Doctoral, fou una revisió bibliogràfica sobre els efectes del foc en les propietats del sòl publicada en el "Boletín de la AGE" (Asociació de Geògrafs d'Espanya) i titulada, "*Efectos de los incendios forestales en la vegetación y el suelo en la cuenca mediterránea: revisión bibliográfica*". Una vegada contextualitzada la investigació, es va procedir a realitzar estudis i experiments concrets. En el primer, els resultats del qual estan publicats en premsa a la revista Catena amb el títol de "*Spatial and temporal variations of water repellency and probability of its occurrence in calcareous Mediterranean rangeland soils affected by fires*", es va estudiar l'evolució de la repel·lència a l'aigua en el sòl immediatament després d'un incendi forestal i en conques cremades 10 i 20 abans en un ecosistema mediterrani de sòl calcari. Es van examinar a més les variacions mensuals degudes als canvis en la humitat del sòl i les variacions espacials baix de diferents tipus de vegetació i en micro parcel·les baix d'un mateix individu i, es va verificar que les cendres afecten a la repel·lència a l'aigua del sòl després d'un incendi. També es calcularen les probabilitats de trobar repel·lència a l'aigua. A partir d'aquest experiment s'obriren noves qüestions sobre si les cendres podien ser repel·lents i com que no existien publicacions sobre aquest tema, aquest fou l'objectiu de la següent part d'aquest treball. Els resultats estan publicats en la revista Geoderma amb el títol "*The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content*". Mitjançant experiments de laboratori també s'analitzà la relació de la repel·lència a l'aigua de les cendres amb el contingut de carboni en el seu color i com afecta la incorporació de les cendres a la repel·lència a l'aigua del sòl. Arribat a aquest punt, es va veure necessari a més, conèixer el seu efecte en la repel·lència a l'aigua del sòl quan les cendres el cobreixen. Per tant, es va procedir a la realització de diversos experiments més, intentant aportar alhora informació sobre el seu paper en la generació d'escolament, en la composició química d'aquestes i en l'erosió produïda en diferents sòls, quan diferent cendres cobreixen diferents sòls i amb capes d'espessor variat. Els resultats obtinguts dels experiments de laboratori estan acceptats per a la seua publicació en la revista Geoderma amb el títol "*Hydrological effects of a layer of vegetation ash on underlying wettable and water repellent soil*", mentre que els obtinguts dels experiments de camp estan publicats en les Actes del la III Conferència Internacional FESP "*Fire effects on soil properties*"en Guimaraes (Portugal) titulat "*Runoff rates, water erosion and water quality from a soil covered with different types of ash*".

Aquesta Tesi Doctoral aporta al coneixement científic una sèrie de contribucions entre les quals destaquen les següents:

1) En quant a les **variacions espacials i temporals de la repel·lència a l'aigua** en sòls calcaris mediterranis i la predicció de la probabilitat de què estiga present en el sòl:

- La repel·lència a l'aigua pot ser una propietat comuna dels sòls calcaris dels boscos mediterranis. En les zones estudiades, la presència de repel·lència a l'aigua del sòl en superfície va oscil·lar estacionalment entre 40-95% amb algunes mostres manifestant repel·lència extrema, però disminuint en profunditat. Tanmateix, es trobaren increments en la repel·lència del sòl a 1 cm de profunditat després d'un incendi forestal, encara que es reduïren a nivells molt baixos després d'un any tant en superfície com en profunditat. Aquests canvis poden ser deguts al llavat de substàncies hidrofòbiques i a la manca de deposició de material hidrofòbic fresc per la poca coberta de vegetació deguda a l'incendi, juntament amb el recobriment de la superfície del sòl amb sediments producte de l'erosió post-incendi compostos de sòl i cendres hidrofíliques. Aquests resultats indiquen que el restabliment de la repel·lència a l'aigua als nivells d'abans de l'incendi pot tardar més de 10 anys segons els resultats d'aquesta investigació a la Serra d'Enguera (València).

- La variabilitat de la repel·lènciaa l'aigua del sòl baix *Pinus halepensis, Quercus coccifera* i *Rosmarinus officinalis*en la mateixa zona d'estudi de 1 és igual d'àmplia que entre individus de la mateixa espècie i dintre de una mateixa parcel·la de 10 cm × 10 cm. La menor variabilitat s'ha mesurat en sòls nuus on la majoria de les mostres foren hidrofíliques. La repel·lència l'aigua també és variable segons la humitat del sòl, essent major en condiciones seques d'estiu.

- A partir de covariances i factors fixes simples de quantificar (tipus de vegetació, profunditat del sòl i humitat del sòl) es va desenvolupar un model estadístic que permet estimar la probabilitat de trobar repel·lència a l'aigua en el sòl per a diferents estacions i condicions meteorològiques en boscos mediterranis madurs. El model obtingut pot representar eina útil encara que es requereix un major estudi i comprovacions per a poder aplicar-lo en altres zones.

2) En referència a les **propietats de les cendres** estudiades:

- Les cendres procedents d'incendis forestals poden ser repel·lents a l'aigua. La major presència i persistència de repel·lència l'aigua en cendres d'incendis de boscos de *Pinus halepensis* i vegetació associada es troba en les mostres recollides en incendis de baixa severitat i, de les produïdes en el laboratori en les generades entre 200-300 °C, mentre que a més de 400°C, la repel·lència desapareix. A més, les cendres produïdes a partir de *Quercus coccifera* i *Pinus halepensis* resulten en majors nivells de presència i persistència de repel·lència a l'aigua que les de *Rosmarinus officinalis*. La repel·lència a l'aigua en les cendres es redueix després de ser humides. Aquesta propietat està relacionada amb la proporció de carboni orgànic/inorgànic i el tipus de components orgànics. Tanmateix, segons els nostres resultats el color no és un bon indicador tant de la repel·lència a l'aigua com de la quantitat de carboni orgànic si es comparen cendres de diferents incendis forestals.

- La quantitat de carboni orgànic total en les cendres en aquest estudi oscil·la entre 4,6% i 31,1% i, en les cendres produïdes en el laboratori entre 22,9% i 60,3%. Els majors valors de carboni orgànic en les cendres generades en el laboratori pot ser degut a que el procés de combustió produït no es directament comparable al d'un incendi forestal.

3) Respecte a la **longevitat de les cendres en el sòl**, s'ha comprovat que una capa de cendres de més de 5 mm recollides en un incendi de baixa severitat, continua recobrint el sòl després de dos simulacions de pluja de laboratori de 82 mm h^{-1} durant 40 min amb una pendent de 10º. Igualment, espessors de més de 5 mm de cendres produïdes en el laboratori a elevades temperatures mantenen el sòl cobert després de dos pluges simulades en el camp de 55 mm h^{-1} durant 1 hora amb una pendent de 5º. Espessors de menys de 5 mm deixen el sòl descobert després d'aquests episodis de pluja.

4) Quant a l'efecte de les cendres en la **repel·lència a l'aigua del sòl**:

- Si les cendres cobreixen el sòl, una capa de cendres hidrofíliques de més de 5 mm d'espessor redueix considerablement la repel·lència l'aigua del sòl ja que incrementa la pressió hidràulica i el contacte entre l'aigua i el sòl, i per tant promou fluxos preferencials al llarg del perfil del sòl. Després de la primera pluja i en condicions de sòl humit, les cendres no produeixen una reducció de la repel·lència l'aigua del sòl significativa.

- Si les cendres estan incorporades en el sòl, poden augmentar o reduir la repel·lència a l'aigua del sòl segons la seva natura hidrofílica o hidrofòbica. Si és el cas de cendres hidrofòbiques, la repel·lència resultant pot ser elevada si s'incorporen al sòl mitjançant mecanismes sense aigua, com l'acció eòlica o la bioturbació.

5) Respecte als efectes d'una capa de **cendres** en la **resposta hidrològica**:

- Les cendres poden augmentar el flux superficial si la conductivitat hidràulica de les cendres ($K_{cendres}$) es menor que la intensitat de la pluja, cosa que pot ocórrer degut a la compactació i encrostament de les cendres quan tenen un alt contingut en carbonat càlcic o si les cendres són repel·lents a l'aigua.

- En tots els altres casos, quan $K_{cendres}$ > intensitat de la pluja, i sobretot per a la primera pluja després d'un incendi, les cendres retarden el començament del flux superficial proporcionalment a l'espessor de la capa degut a la seva elevada capacitat d'emmagatzemar aigua.

- Una vegada les cendres estan saturades, es produeix un flux superficial i subsuperficial entre les cendres i el sòlfins que ambdues capes s'equilibren i passa a dominar el procés la capacitat d'infiltració del sòl.

- Tanmateix, la capa de cendres pot modificar la capacitat d'infiltració del sòl (a) incrementant-la respecte un sòl no cobert de cendres perquè prevenen el sòl d'encrostar-se i perquè poden reduir la seua repel·lènciaa l'aigua(veure punt 4), o (b) reduint-la si taponen els porus del sòl, que dependrà de la mida de les partícules d'ambdós materials.

6) Les **cendres protegeixen el sòl** de l'erosió produïda per l'impacte de les gotes de pluja en tots els casos i prevenen l'erosió laminar sempre que no es saturen i no es produesca flux superficial. Aquesta reducció és especialment important sobre sòls repel·lents nuus després d'un incendi, que són els que solen produir les majors taxes d'erosió. El sediment produït en sòls coberts per cendres està associat al flux superficial generat (veure punt 5), però durant episodis de pluges intenses, les cendres poden augmentar la capacitat de transport del sediment i contribuir a episodis extrems de erosió.

7) Els **nutrients** alliberats per les cendres modifiquen la qualitat de l'aigua d'escolament, incrementant el seu pH, la conductivitat elèctrica i especialment la quantitat de cations. Els nutrients solubilitzats no són exactament els mateixos que els existents en la composició original en forma de cendres i depenen del volum de l'escolament produït (veure punt 5), de la solubilitat dels compostos de les cendres i de les interaccions químiques amb el sòl. Després de la primera pluja, si aquesta és abundant, la major part dels nutrients són solubilitzats i lixiviats o arrastrats amb el flux superficial, però alguns d'ells poden augmentar unes setmanes després en produir-se canvis químics tant a les cendres com al sòl.

1. Introducción general

1. Introducción general[1]

1.1. Los incendios forestales en la cuenca mediterránea

El fuego tiene un papel importante en los procesos que controlan el Sistema Tierra. La abundancia de carbones en los sedimentos demuestra que ha formado de parte de los procesos terrestres desde principios del Devónico (hace 400 millones de años), después de que las plantas colonizaron la Tierra. Desde entonces, su frecuencia e intensidad ha ido cambiando según los niveles de oxígeno atmosférico y el clima (Scott, 2000a; Pyne, 2001).

Como componente del sistema terrestre, el fuego ha influido en los ciclos biogeoquímicos de ciertos elementos como el oxígeno, carbono o fósforo entre otros, por tanto en la composición de los gases de la atmósfera y a su vez en el clima de la Tierra (van der Werf et al., 2006; Lenton, 2001; Scott, 2009); en las geoformas, el transporte de materiales y las tasas de sedimentación (Shakesby y Doerr, 2006; Scott, 2009); y especialmente tiene una íntima relación entre el tipo de vegetación y distribución de los biomas de la Tierra. Según Bond y Keeley (2005), la distribución de los biomas de la tierra no se debe solo a factores climáticos ya que hay ecosistemas de sabana, pradera o incluso zonas de matorral mediterráneo que poseen un clima en el que potencialmente podrían sostener un bosque. Es decir, el tipo de vegetación existente en un cierto ecosistema se debe también a la recurrencia del fuego con un régimen concreto en cuanto a severidad y frecuencia (Bond y Keeley, 2005; Lloret y Zedler, 2009; Pausas y Keeley, 2009).

Los ecosistemas mediterráneos poseen un clima de transición entre el régimen templado oceánico y el tropical seco. Su característica más importante es la coincidencia de la época seca con la cálida. Esto es porque en verano, el Mediterráneo se encuentra bajo la influencia de los anticiclones subtropicales secos que se retiran en invierno hacia latitudes más bajas y dejan paso a las borrascas atlánticas de latitudes templadas. Esta peculiaridad propicia una vegetación angostada en verano y una gran acumulación de ramas y hojas muertas debido a las bajas tasas de descomposición y, por tanto una gran susceptibilidad a los incendios forestales (Arianoutsou et al., 1993; Pérez Cueva, 1994). No se sabe con exactitud cuál era el régimen de incendios "natural", pero debió caracterizarse por incendios también en verano tras tormentas

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[1] La primera mitad de la introducción (sección 1.1. y 1.2) es una versión adaptada del artículo "Efectos de los incendios forestales en la vegetación y el suelo en la cuenca mediterránea: Revisión bibliografica" de Merche B. Bodí, Artemi Cerdà, Jorge Mataix-Solera y Stefan H. Doerr, publicado en el Boletín de la AGE 58, 33-55 (2012). Ver anexo 6.1.

eléctricas, menos frecuentes que en la actualidad y con áreas afectadas más grandes, ya que el fuego actuaba sobre un paisaje más homogéneo que el actual y sin barreras antrópicas como carreteras, ciudades, pastos o campos de cultivo. La mayoría de incendios debieron ser de copa, si bien en algunas zonas pudieron ser de superficie como demuestra el que en los bosques mediterráneos de *Pinus nigra* y *Pinus sylvestris* la vegetación está adaptada a ese tipo de incendios que además ocurren con menos frecuencia (Naveh, 1975; Arianoutsou et al., 1993; Fulé et al., 2008; Pausas et al., 2008).

El fuego ha determinando la evolución de las formaciones vegetales por medio de la adaptación y selección natural. Los principales rasgos seleccionados en la evolución de la vegetación son los que guardan relación con la regeneración, característica fundamental para la persistencia en ambientes con incendios frecuentes (Arianoutsou et al., 1993; Trabaud, 1994; Pausas, 2010). Los mecanismos que se observan en la cuenca mediterránea son la capacidad de rebrotar tras el fuego (por ejemplo: *Quercus ilex*, *Quercus coccifera, Juniperus oxycedrus, Brachypodium retusum, Ampelodesmos mauritanica, Chamaerops humillis*), y de reclutar nuevos individuos a partir de un banco de semillas acumulado (en el suelo o en la copa) resistente al calor de fuego. En algunos casos el calor rompe la dormancia de las semillas (en especies con semillas duras e impermeables como *Ulex parviflorus* o *Cistus Albidus*), en otras el humo estimula la germinación (como la *Erica Umbellata* o *Lavandula sp*.); y, en las especies serótinas (es decir, con banco de semillas aéreo como el *Pinus halepensis* o *Pinus Pinaster*) el calor estimula su dispersión. Mediante estos procesos, las poblaciones se restablecen rápidamente en los espacios abiertos que los incendios generan y a menudo aumentan su tamaño poblacional temporalmente respecto a las condiciones previas al incendio. No obstante, existen otras especies que carecen de estrategias de adaptación al fuego. Algunas de ellas con elevada producción de semillas y mecanismos de dispersión eficientes recolonizan el área quemada con prontitud desde fuera del perímetro del incendio y otras lo hacen lentamente, apareciendo en espacios donde el fuego ha estado ausente durante tiempo (por ejemplo: *Juniperus phoenicia*, *Cedrus sp*., *Pinus nigra*, *Pinus sylvestris*). Además de los mecanismos de regeneración, hay otras adaptaciones al fuego como pueden ser la resistencia pasiva frente al calor por un elevado contenido en agua (plantas suculentas) o por su gruesa corteza como el *Quercus suber,* y la elevada inflamabilidad para facilitar la propagación del incendio con aceites, resinas o estructura de las hojas y ramas (por ejemplo *Erica multiflora*, *Pinus halepensis* o *Quercus ilex*) (Arianoutsou et al., 1993; Crosti et al., 2006; Pausas et al., 2008).

Una vez adquirido el control del fuego por los humanos, se utilizó para numerosas actividades domésticas (como cocinar, producir luz o calentarse) y para adquirir recursos del entorno. El fuego se empleaba para abrir claros en el bosque y crear zonas más accesibles, favorecer la producción de ciertos alimentos, crear zonas de cultivo y pastos, fertilizar con las cenizas, controlar plagas y malas hierbas, propiciar mejores zonas de caza y luchar entre grupos tribales (Naveh, 1991). Estas quemas practicadas por los humanos y la fragmentación del paisaje (y del combustible) debido a la expansión de la agricultura y las sociedades, hicieron que el régimen de incendios fuera cambiando a lo largo de la historia. En algunos sitios y momentos, disminuyó la frecuencia; en otros, aumentó (Naveh, 1974; Carcaillet et al., 2002; Pausas, 2010). En general, la frecuencia de incendios de origen antrópico se incrementó a partir del 6000 BP, durante el Neolítico, que junto con un régimen climático más seco, forzó un cambio en la vegetación los bosques mediterráneos. Análisis palinológicos y de carbón coinciden que en ciertas partes se produce la expansión de *Quercus* perennifolios (Carrión y Dupré, 1996; Colombaroli et al., 2007; Sadori y Giardiani, 2007), pero en otras partes del Mediterráneo, esta presión antrópica de incendios produce un incremento del área de matorral y de praderas (Colombaroli et al., 2008). Sin duda, el bosque mediterráneo y su régimen de incendios está lejos de ser natural debido a la larga acción humana sobre el medio natural (Naveh, 1975).

Los incendios controlados se han seguido practicando hasta los años 60 en España y en la Europa mediterránea, lo que significa que hasta mediados del siglo XX se garantizaba la gestión del monte mediante un aprovechamiento sistemático, y muchas veces abusivo, de los recursos forestales del territorio. Los recursos forestales se explotaban para obtener leña para la lumbre y cocina, madera y pasto. Además se roturaba el suelo o se quemaba la vegetación para sembrar cereales, plantar frutales, viñas, u olivos. El resultado era un paisaje compartimentado y diverso con bajo riesgo de incendios, los cuales eran rápidamente sofocados por una población que vivía en el monte y del monte (Molinero et al., 2008). No obstante, en algunas ocasiones también se producían incendios incontrolados en las grandes masas forestales que quedaban, como el de la Font Roja en Alcoy (Alicante) en 1840 (Montiel Molina, 1994), si bien el riesgo era menor debido al uso y abuso de los recursos forestales.

En los años 60, la industrialización y terciarización de la economía española desembocaron en un masivo éxodo rural que dio lugar a un cambio de los usos del suelo que se cristalizó en el abandono de los campos de zonas de montaña y un incremento de la cubierta vegetal. En la actualidad, estas zonas no existe apenas explotación ni se obtiene beneficio directo, ni siquiera para la gente que aún permanece en las zonas rurales, ya que nuevas normas y leyes restringen las talas, zonas de pasto y caza (Molinero et al., 2008). A este cambio de usos del suelo han contribuido también las repoblaciones de pinar, especialmente de *Pinus halepensis*, la política de supresión de incendios y el aumento de nuevas zonas residenciales en la interfaz urbano-forestal (Pausas et al., 2008; Vallejo et al., 2009). Las consecuencias han sido un aumento de los incendios forestales sin control desde 1970 (Figura 1) y un cambio en el régimen de incendios que dentro de su carácter antrópico, antes dependía principalmente de la disponibilidad de vegetación o combustible que hubiera y, ahora que la biomasa es abundante, depende de unas condiciones climáticas favorables. El fuego se producirá entonces a pesar del esfuerzo en pistas forestales, cortafuegos, infraestructuras y medios de extinción (Pausas y Fernández-Muñoz, 2011).

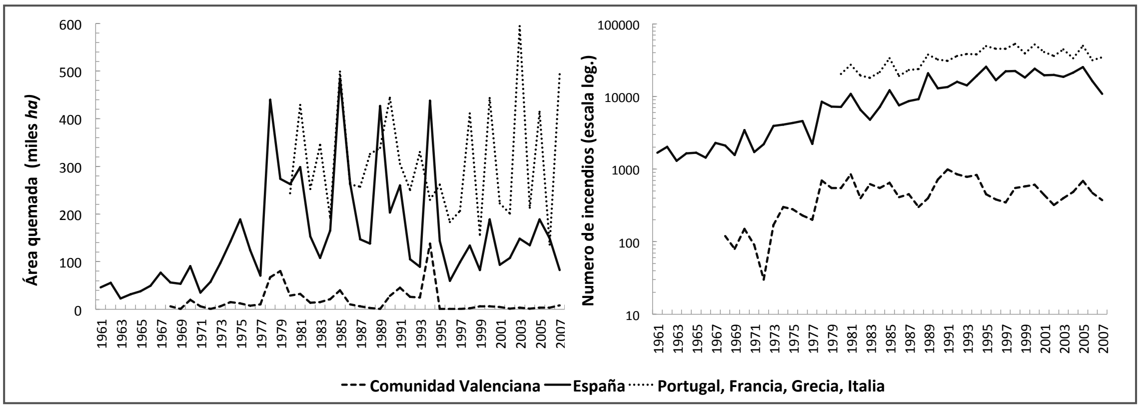

Figura 1. Izquierda: Superficie (miles de ha) afectada por incendios forestales en la Comunidad Valenciana, España y otros países de la Europa mediterránea (Portugal, Francia, Italia y Grecia). Derecha: Número de incendios forestales anuales en la Comunidad Valenciana, España y otros países de la Europa mediterránea (Portugal, Francia, Italia y Grecia). Fuente: "European Comission, Institute for Environment and Sustainability" y "Generalitat Valenciana, Conselleria d'Infraestructures, Territori i Medi Ambient".

Los incendios forestales, a diferencia de las quemas controladas, ponen en peligro propiedades y vidas humanas. También pueden suponer un riesgo para el ecosistema si ocurren con mayor intensidad o con mayor frecuencia. Aunque los ecosistemas mediterráneos son capaces de convivir con él este cambio de régimen puede provocar efectos adversos (Pausas y Keeley, 2009). Si el fuego afecta al ecosistema forestal, el riesgo de crecidas de caudales hídricos y de erosión aumenta, y se pueden llegar a producir problemas de inundaciones e incluso en el suministro de agua potable (Cerdà y Robichaud, 2009b), sobre todo en la franja climática mediterránea en la que a la estación seca de incendios le sigue la época de lluvias normalmente torrenciales. Pero los incendios forestales no son siempre eventos catastróficos, los de baja intensidad son de pequeño impacto y promueven la vegetación herbácea, de nutrientes y aclaran los bosques, lo que promueve un hábitat más sano y heterogéneo (Neary et al., 1999). Éste fue el incendio controlado del matorral mediterráneo durante milenios. Es por tanto necesario estudiar y analizar cada caso concreto para poder decidir si el incendio ha causado o no daños en el ecosistema.

Las políticas forestales tradicionales en la cuenca mediterránea han sido la de replantar bosques monoespecíficos de pino tras cualquier incendio (incluso años más tarde), a pesar de que algunos son muy inflamables (Pausas et al., 2008). Además, estas plantaciones se realizan mediante maquinaria pesada que puede provocar más daño al suelo que el propio incendio, y obviamente, estas actuaciones son más costosas económicamente que dejar que la vegetación se recupere por ella misma. La decisión de dónde y cuándo usar tratamientos de remediación después de un incendio requiere una evaluación de la severidad de incendio, el clima, suelos, topografía e hidrología de la cuenca (Robichaud, 2009). Las actuaciones que siguen a un incendio son el acolchado con restos vegetales e hidrosiembras, repoblaciones con diferentes tipos de semillas y barreras de erosión. Para las repoblaciones en la cuenca mediterránea se recomienda la introducción de especies rebrotadoras de hoja ancha para generar una mayor diversidad de especies y para mejorar la resiliencia del ecosistema, incrementando su heterogeneidad (Pausas et al., 2004; Vallejo et al., 2006). No obstante, estos tratamientos postincendio son muy caros y solo deben ser aplicados si el riesgo de degradación del suelo y vegetación es elevado. Por ello, es justificable y efectiva la no actuación tras un incendio, tanto por razones económicas como medioambientales. Un buen ejemplo de ello lo encontramos en los casos en los que las hojas caídas de los pinos después de un incendio actúan como acolchado natural (Figura 2), o en los que la propia regeneración de la vegetación es suficiente para reducir las tasas de erosión (Cerdà y Doerr, 2008).

Figura 2. Acolchado natural que se produce cuando en ocasiones parte de las acículas del pino no han sido destruidas directamente por el fuego y con el paso del tiempo van cayendo al suelo. Incendio de Gorga de Julio de 2012. Foto realizada en Septiembre 2012 por Jorge Mataix-Solera.

Respecto a la prevención, hasta ahora las políticas forestales han optado por una estricta supresión de los incendios forestales y divulgan sólo sus efectos negativos, de manera que la sociedad ha llegado a considerar que el fuego debe suprimirse completamente. Pero como se ha comprobado, esta política propicia los incendios de alta intensidad, precisamente los que desencadenan procesos de degradación más intensos. Ejemplos son las olas de grandes incendios como los de 1979 y 1994 en la Comunidad Valenciana y, más recientemente las de Galicia en 2006, Grecia en 2007 o el sureste de Australia en Febrero de 2009, lo que demuestra que no podemos excluir el fuego de unos ecosistemas que han convivido con él tanto de manera natural, como bajo la gestión del hombre. Es por tanto necesaria una gestión del monte que puede incluir desbroces, cultivo, pastoreo o incluso en la que se contemple la posibilidad de usar el fuego controlado para a su vez combatir los incendios descontrolados, opción que ya se aplica en Canarias y Cataluña, y en países como EEUU o Australia.

1.2. Efectos de los incendios forestales

Los efectos del los incendios forestales son perceptibles a distintas escalas. Por ejemplo, a escala global, existe relación entre los efectos del fuego, el ciclo del carbono y el cambio climático. Se considera que la gran parte de las emisiones de CO_2 de la deforestación y degradación de bosques proviene de incendios forestales, especialmente de aquellos ecosistemas no adaptados donde el reemplazo vegetal no ocurre rápido, como pueden ser en las selvas tropicales o en la turba (Le Quére et al., 2009; van der Werf et al., 2009); pero a la vez, las formas de carbón derivadas de la combustión tras los incendios son una de las formas más resistentes a la degradación y constituyen un sumidero de carbono (Schmidt y Noack, 2000). No obstante, los cambios más notables que produce un incendio forestal son a escala del propio ecosistema. El fuego consume la vegetación, y modifica las propiedades del suelo, afecta a la micro y macrofauna, a los procesos hidrológicos y geomorfológicos y a la calidad de las aguas (DeBano et al., 1998; Cerdà y Robichaud, 2009b).

Los efectos del fuego son muy variados debido a los múltiples factores de los que depende la severidad del incendio: intensidad (definida como la velocidad de liberación de energía durante el proceso de combustión), duración, tipo de combustión, área quemada, topografía, tiempo atmosférico, tipo de vegetación (viva o muerta) y tipo de suelo, humedad edáfica, vegetal y atmosférica y tiempo desde el ultimo incendio (Neary et al., 1999; Keeley, 2009). Con todas éstas variables, incluso en un mismo incendio la severidad del fuego varía y suele resultar en un mosaico irregular de zonas mas o menos afectadas (Figura 3). No obstante, la capacidad del ecosistema para recuperarse no sólo depende de la severidad del fuego sino también de la resiliencia y adaptaciones del propio ecosistema al fuego (Keeley, 2009). Los problemas de recuperación suelen ocurrir cuando se producen incendios de elevada severidad o cuando el régimen de incendios de un cierto ecosistema varía, ya sea la frecuencia, extensión afectada, estación en que ocurre o tipo de incendio (de copas, de superficie o subsuperficial; The Nature Conservancy, 2004).

Figura 3. Izquierda: Incendio de moderada severidad en Pinoso (Alicante), donde algunos de los *Pinus halepensis* no están completamente quemados pero hay manchas de ceniza blancas que indican la completa combustión de algunos matorrales. Derecha: Incendio de alta severidad en la sierra de Enguera (Valencia) donde las copas de los *Pinus halepensis* están completamente quemadas.

1.2.1. Efectos de los incendios forestales en las propiedades físicas y químicas del suelo

El suelo es un componente básico del ecosistema forestal (Mataix-Solera y Guerrero, 2007; Waring y Running, 2007). Sus propiedades físicas, químicas y biológicas están íntimamente relacionadas y los cambios producidos en alguna propiedad por una perturbación, en este caso por el fuego, van a afectar a las otras. Estos cambios son diferentes según el tipo de suelo y pueden ser reversibles a corto o largo plazo o permanentes, pero la recuperación de todas sus funciones, químicas, físicas y biológicas son vitales para el restablecimiento de los demás elementos del ecosistema (Neary et al., 1999; Certini, 2005).

Los cambios producidos por el fuego no son continuos en el suelo debido a la irregularidad de la severidad del fuego y, se producen principalmente en los primeros centímetros. El componente de la severidad que más afecta al suelo y a sus propiedades es la duración del fuego. Incendios rápidos en biomasa fina como en la sabana pueden liberar gran cantidad de energía y llegar alcanzar altas temperaturas en la llama, pero no transmiten tanto calor al suelo como un incendio lento en combustibles compactos y densos, como por ejemplo un tronco caído (Neary et al., 1999; Certini, 2005). Así, los cambios en las propiedades del suelo se deben fundamentalmente a la combustión y calentamiento (cambios directos) y también a la situación microclimática después de la pérdida de la cubierta vegetal y recubrimiento de las cenizas (cambios indirectos). Aunque en algunos casos los efectos directos en indirectos son difíciles de separar, en la sección 1.3.2. se tratan sobre los efectos los efectos de las cenizas en el suelo (Figura 4 izquierda; Raison y McGarity, 1980a; Neary et al., 1999; Mataix-Solera y Guerrero, 2007).

Los cambios en el carbono orgánico del suelo son complejos y variados según la severidad del incendio. Además, estos cambios pueden tener repercusiones en otras propiedades del suelo como la capacidad de intercambio catiónico o la estabilidad de agregados (Neary et al., 1999). En incendios de baja severidad puede haber un incremento de la materia orgánica, en cambio a intensidades elevadas la cantidad de materia orgánica de la superficie del suelo disminuye por oxidación (Giovannini, 1994; Mataix-Solera et al., 2002; González-Pérez et al., 2004). Según Neary et al. (1999), en un incendio severo con temperaturas de 675 ºC en la superficie del suelo se destruye totalmente la materia orgánica en la superficie y parcialmente hasta 25 mm. Pero el fuego no sólo modifica la cantidad de la materia orgánica, también altera su calidad. Actúa como un agente que acelera las tasas de mineralización del carbono orgánico y además modifica las tasas de descomposición postincendio ya que, a medida que se incrementa la temperatura, el humus sufre modificaciones que la hacen más resistentes a la degradación microbiana, llamado humus piromórfico (Knicker, 2007; González-Vila et al., 2009). Ésta materia orgánica carbonizada que se produce en grandes cantidades y se acumula en el suelo, puede contribuir en un 30-40% al carbono del suelo en ecosistemas propensos a incendios forestales y al secuestro de carbono a largo plazo, siendo un componente significativo en el ciclo global del carbono (Forbes et al., 2006).

El pH y la conductividad eléctrica del suelo normalmente aumentan como resultado de la desnaturalización de los ácidos orgánicos y la liberación de componentes inorgánicos de la combustión de la materia orgánica. Es por tanto que los incrementos significativos ocurren a temperaturas mayores de 450-500 ºC coincidiendo con la completa combustión de la materia orgánica y de liberación de bases (Giovannini et al., 1990a; Certini, 2005). No obstante el mayor incremento se debe al aporte de carbonatos, cationes básicos y óxidos procedentes de las cenizas. El tiempo de recuperación del pH inicial es variado y se considera que es más o menos rápido según el tiempo que las cenizas permanezcan en el suelo y a la capacidad tampón de cada tipo de suelo (Certini, 2005; Mataix-Solera y Guerrero, 2007).

El fuego también produce un descenso en el nitrógeno (N) del suelo debido a su bajo punto de volatilización (< 200 ºC) (Raison et al., 1985). No obstante, se han observado aumentos del N inorgánico como el amonio (NH_4^+) a temperaturas mayores de 200 ºC, cuando ya se han iniciado los procesos de mineralización físico-química por combustión. Además, después de algunas semanas con las nuevas condiciones de temperatura y humedad tras el incendio, las tasas de la concentración de nitratos (NO_3^-) suele aumentar debido a la nitrificación del amonio (Christensen, 1973; Khanna y Raison, 1986; Mataix-Solera y Guerrero, 2007; Raison et al., 2009). Asimismo, en los meses posteriores se produce un aumento del N orgánico por microorganismos promovidos por el incendio y especies de leguminosas fijadoras como el *Ulex parviflorus* (Neary et al., 1999; Raison et al., 2009). Al igual que el N, el calentamiento convierte el fósforo (P) orgánico en ortofosfato, una forma asimilable del P por la biota. Además, en el caso del P su punto de volatilización es mayor que el del N, por lo que no se pierde tanto por este medio (Certini, 2005). El aporte de cenizas de la vegetación también enriquece el suelo con un aumento de cationes (Ca^{2+}, Mg^{2+}, K^+, Na^+) que según Kutiel y Naveh (1987a) es considerado el mayor factor de crecimiento de la vegetación en los ecosistemas mediterráneos. Sin embargo, normalmente la capacidad de intercambio catiónico se ve reducida tras un fuego intenso, proporcionalmente a la pérdida de materia orgánica y por tanto, parte de los nuevos nutrientes no podrán ser retenidos (Carballas, 1997) y si las condiciones ambientales son húmedas, se perderán por erosión hídrica, eólica o lixiviación (Lasanta y Cerdà, 2005; Gimeno-García et al., 2000; Raison et al., 2009). La fertilización postincendio puede ser efímera, y durar de 4 a 5 meses (Gimeno-García et al., 2000; Kutiel y Naveh, 1990), hasta 14 meses (Kutiel y Naveh, 1987a) o incluso 7 años (Úbeda et al., 2005). Una adecuada recuperación de la vegetación favorecerá que esa fertilización efímera se traduzca en la captación de esos nutrientes, o de lo contrario la zona se empobrecerá.

En cuanto a las propiedades físicas, el cambio más fácil de reconocer es el color. El suelo se ennegrece cuando las temperaturas oscilan entre 100-250 ºC y cuando se alcanzan mayores temperaturas cambia a un color más rojizo debido a la formación de óxidos de hierro (Certini, 2005). En este suelo desprovisto de vegetación y ennegrecido por el calor y las cenizas se han medido aumentos medios de temperatura (Mataix-Solera, 1999; Iverson y Hutchinson, 2002) y mayores amplitudes térmicas diarias hasta 3 años después (Massman et al., 2008).


Figura 4. Izquierda: Capa de cenizas que cubre el suelo después de un incendio en Pinoso (Alicante, Julio de 2009), factor que produce efectos indirectos del incendio en el suelo. Derecha: Efecto directo del incendio en el suelo. Fusión térmica del suelo habitual en suelos con óxidos y hidróxidos de hierro en su fracción arcilla (Mataix-Solera y Guerrero, 2007).

La textura también puede verse modificada en incendios de alta intensidad debido a una disminución del contenido de arcilla por fusiones térmicas de sus óxidos y hidróxidos de hierro (Figura 4, derecha; Giovannini et al., 1990b), pero también por la pérdida de finos en caso de que haya erosión hídrica posterior (Llovet et al., 1994). Igualmente la estabilidad de agregados también puede cambiar tras el paso del fuego. En la revisión de Mataix-Solera et al. (2011) sobre efectos de los incendios en la agregación del suelo, se concluye que en incendios de baja severidad no hay demasiados cambios, pero a moderada y alta severidad ocurren diferentes cambios según las propiedades del suelo. Si éste tiene un alto contenido en arcillas y carbonato cálcico la estabilidad puede aumentar debido a fusiones de arcillas que endurecen los agregados, pero si la materia orgánica es la principal responsable de mantener los agregados unidos se producirá una reducción relacionada con su pérdida. La reducción de la agregación del suelo modifica su porosidad (disminuyendo proporcionalmente los poros de mayor tamaño) y la capacidad de retención de agua del suelo (Boix-Fayos, 1997; Neary et al., 1999; Stoof et al., 2010).

La hidrofobicidad o repelencia es una propiedad del suelo que impide que el agua penetre en él. Ésta puede verse provocada, aumentada o disminuida según la repelencia al agua inicial y la temperatura alcanzada en el suelo tras un incendio. A grandes rasgos, si la temperatura alcanzada en el suelo es de 250-400°C (dependiendo de la duración del calentamiento), la repelencia al agua puede aparecer o aumentar, y si es mayor se destruye (Doerr et al., 2009a). Los motivos de la aparición o aumento de la repelencia debido al fuego se deben a la volatilización de los compuestos orgánicos de las plantas y del mantillo y su condensación en la superficie del suelo o a profundidad según el gradiente térmico producido (DeBano et al., 1976). Además, parece ser que el fuego convierte algunas sustancias orgánicas en hidrofóbicas por pirólisis o por alteraciones moleculares de la materia orgánica (Atanassova y Doerr, 2011; Savage et al., 1972). Los factores que principalmente controlan la hidrofobicidad son, junto con la temperatura alcanzada, el tipo de vegetación y la cantidad de hojarasca y la materia orgánica presente, siendo las especies con aceites y resinas (como el *Pinus halepensis* o

Eucalyptus sp.) los mayores inductores de sustancias hidrofóbicas en el suelo (Doerr et al., 2000). Además, los suelos ácidos y de textura arenosa son más propensos a desarrollar repelencia al agua aunque ésta también aparece en suelos calcáreos (Doerr et al., 2000; Mataix-Solera y Doerr, 2004). No obstante Mataix-Solera et al. (2008) ha demostrado que la baja ratio materia orgánica/arcilla y una presencia elevada de caolinita en el suelo denominado comúnmente como *terra rossa* (principalmente *Rhodoxeralfs*), lo hacen muy poco susceptible a desarrollar ésta propiedad. La aparición o aumento de la repelencia al agua del suelo por el fuego puede durar desde pocos meses a años y se reduce debido a la ausencia de nueva vegetación que aporte sustancias orgánicas combinado con el recubrimiento de sedimentos erosionados (Hubbert y Oriol, 2005; Tessler et al., 2008). Por otra parte si desaparece por efectos de las altas temperaturas, la repelencia volverá a restablecerse en ecosistemas donde es natural junto con la recuperación de la vegetación y de nuevos microorganismos (Doerr et al., 2009b; Jordán et al., 2010). La repelencia al agua también cambia rápida y estacionalmente según la humedad del suelo (Keizer et al., 2008; Leighton-Boyce et al., 2005). Existe una "zona de transición" que define un rango de humedad por encima del cual las muestras dejan de ser repelentes, es decir, un suelo repelente pasará a no serlo si sobrepasa el limite de la "zona de transición" y volverá a ser repelente cuando se seque (Dekker et al., 2001).

Con el aumento de la repelencia al agua tras el incendio obviamente también se verán reducidas la conductividad hidráulica y la capacidad de infiltración (Martin y Moody, 2001; Robichaud, 2000). Sin embargo, la reducción de estas últimas no puede deberse únicamente a la repelencia al agua si no también a otros factores propiciados por el fuego como la reducción de la porosidad, la compactación y el encostramiento (Cerdà y Robichaud, 2009a). Todas estos cambios en la propiedades físicas y químicas provocan también un aumento de su erodibilidad (Giovannini et al., 2001).

1.2.2. Efectos de los incendios forestales en la generación de escorrentías y erosión del suelo

Las modificaciones en las propiedades físicas e hidrológicas del suelo tras un incendio, según su severidad y el tipo de suelo (cambios en la cantidad y calidad de la materia orgánica, textura, agregación del suelo, hidrofobicidad, conductividad hidráulica, capacidad de retención de agua, capacidad de infiltración y erodibilidad), unidos a la disminución de vegetación y hojarasca y, si es el caso, a pendientes elevadas y episodios extremos de lluvia, pueden propiciar un aumento de mayor o menor magnitud de la escorrentía superficial y la erosión respecto antes del incendio o zonas similares no quemadas (Kutiel et al., 1995; Cerdà, 1998a; Andreu et al., 2001; Martin y Moody, 2001; La Roca et al., 1988; Larsen et al., 2009). Estos aumentos en las tasas de escorrentía y erosión se han medido en periodos de hasta 3 o 10 años, aunque normalmente vuelven a valores de antes del incendio a la vez que la recuperación de la vegetación. Es por tanto que los primeros 4-6 meses son los más susceptibles, especialmente en la cuenca mediterránea por las intensas lluvias otoñales que siguen al verano (Cerdà y Doerr, 2005; Gimeno-García et al., 2007; Mayor et al., 2007). No obstante, inmediatamente tras el incendio, las cenizas resultantes cubren el suelo formando un sistema de dos capas y los procesos hidrológicos que suceden son distintos a los de un suelo descubierto (Kinner y Moody, 2010). Ese periodo durante el cual es suelo esta cubierto de cenizas ha sido muy poco estudiado por la comunidad científica. En esta Tesis Doctoral es tratado en detalle en la sección 1.3.3. ya que es clave para el trabajo de investigación aquí presentado.

En estudios en la cuenca mediterránea se han detectado el aumento en las tasas de escorrentía en zonas quemadas respecto a las no quemadas a escala tanto de parcela (Cerdà, 1998a; Úbeda y Sala, 1998; Cerdà y Doerr, 2005; Gimeno-García et al., 2007) como de cuenca (Inbar et al., 1998; López y Batalla, 2001; Mayor et al., 2007). Pero normalmente, sólo las tormentas individuales de gran intensidad son las que desencadenan los episodios extraordinarios de arroyada postincendio y/o inundaciones. Hemos presenciado caudales súbitos postincendio en la zona de estudio del Macizo del Caroig (Navalón) dos meses tras un incendio, donde 15 mm en 10 minutos fueron suficientes para desencadenar una súbita arroyada (Figura 5; Cerdà et al., 2009b). Este aumento en los caudales también se ha medido hasta varios años después del incendio si las lluvias son intensas y la recuperación tras el incendio es lenta. Un ejemplo de esta influencia de las zonas quemadas en los caudales han sido las inundaciones en Haifa (Israel) del 12 al 16 de Enero de 2012, 13 meses después de un incendio de alta intensidad y tras una lluvia de 196 mm en tres días, 117 mm de ellos registrados en 24 horas (Naama Tessler, comunicación personal). Durante estas tormentas, las cuencas quemadas generalmente responden más rápido a la lluvia que antes del incendio o que cuencas no quemadas (Llovet et al., 1994; De Luis et al., 2003; Mayor et al., 2007).

Respecto a la erosión de suelo, de acuerdo con la exhaustiva revisión de Shakesby (2011) sobre la erosión postincendio en la cuenca mediterránea, los valores medidos bajo condiciones de lluvia natural oscilan de media el primer año en mediciones sobre un punto entre 45-56 Mg ha^{-1} y en experimentos con parcelas, la mitad de los estudios revisados no superan 1 Mg ha^{-1} y sólo en 5 trabajos se excede de 10 Mg ha^{-1}. Igualmente los valores que se indican en las revisiones de Cerdà y Bodí, (2007) y de Pausas et al., (2008) incluyendo también simulaciones de lluvia, no sobrepasan las 10 Mg ha^{-1}. Las mayores tasas de pérdida de suelo se registran en orientaciones de solana (Marqués y Mora, 1992; Cerdà et al., 1995), en incendios de alta severidad, y tras eventos de precipitaciones de elevada intensidad, que son los que desencadenan los episodios extraordinarios de erosión. En territorios más lluviosos, además de la intensidad, el volumen de precipitación se ha mostrado como un factor relevante (Cerdà y Lasanta, 2005).

Las tasas de erosión tras incendios forestales en el mediterráneo son bajas comparándolas con las de Estados Unidos o Australia, que varían entre 2,5 y 197 Mg ha^{-1} $año^{-1}$ (ver revisión de Shakesby y Doerr, 2006). También son menores que las registradas en el mediterráneo en otros usos del suelo como los taludes de carreteras, pistas forestales o zonas cultivadas, sobre todo cuando el laboreo es intensivo y con herbicidas (utilizando similares métodos de medida y pendientes; Cerdà y Bodí, 2007; Cerdà et al., 2009a; García-Orenes et al., 2010; Shakesby, 2011). El motivo de que las tasas de erosión sean bajas puede deberse a que el suelo es somero a causa del uso esquilmante por distintas sociedades agrícolas. La elevada pedregosidad de los suelos mediterráneos en general es un indicador de estas intensas pérdidas de suelo y, que a la vez lo protege de mayor erosión (Poesen et al. 1994; Cerdà, 2001a). Es por tanto que a pesar de que la cantidad de suelo erosionado no sea tan elevado como en otros lugares o usos del suelo, la pérdida pueda ser relevante, sobre todo en zonas con incendios recurrentes. Además, la capa superficial del suelo susceptible a erosionarse contiene la mayoría de los nutrientes que se pueden perder con solo unos milímetros de suelo. Tanto Cerdà (2001b) como Verheijen et al. (2009) apuntan que "aunque la tasa de formación de suelo está todavía por determinar, tasas de erosión de más de 1 Mg ha^{-1} $año^{-1}$ son insostenibles en la cuenca mediterránea".

Figura 5. Movimiento de sedimentos en una cuenca de la Sierra de Enguera (Valencia) en la primera tormenta de 15 mm en 10 min dos meses y medio tras un incendio.

1.3. Ash definition, properties and its interactions in the fire affected ecosystem

The scientific research in the field of wildfire effects has paid relatively little attention to ash until now. Studies to date have focused mainly on the effects of fire on vegetation recovery, soil properties such as water repellency, or runoff generation and erosion, all of which tend to be more readily evident and accessible to quantify and characterise than ash. Ash is often rapidly redistributed and removed from burnt sites by wind and water erosion (Cerdà and Doerr, 2008; Larsen et al., 2009), which typically occurs before the commencement of post-fire field studies, so that the potential importance of ash may not always be obvious. Another reason might for the limited attention to date may be the multi-disciplinarity required for the ash study. Ash may be considered separately from soil, vegetation, charcoal or from the cycle of some nutrients. However, now it is recognised that ash is related with all these elements and cycles and therefore needs a broader view of study, adequate for the wide field of the Physical Geography and Soil Science.

1.3.1. Ash definition

During a fire, the organic material of the forest is subjected to a combustion process, which is an oxidation chemical reaction. This reaction must start with an ignition and needs oxygen supply (O_2), resulting in a process that modifies the nature of the original organic material and yields particulate and gaseous by-products and heat. For instance, the combustion reaction of pure plant-synthetized cellulose as a fuel is:

$$\text{ignition} + (C_6H_{10}O_5)_n + O_2 \Rightarrow CO_2 + H_2O + \text{heat}$$

with CO_2 and H_2O being reaction by-products. Depending on the combustion completeness, other organic by-products can be yielded as charcoal, mineral ash and organic volatiles or gases, for instance carbon monoxide (CO), methane (NH_4^-) or nitrogen dioxide (NO_2; DeBano et al., 1998).

The particulate and gaseous by-products are new compounds regarding their physical and chemical properties and have been given different terms depending on their nature (Figure 6). Jones et al. (1997) proposed a terminology for fire-altered matter and make a distinction between ash, charcoal, partially charred material, soot and gases and considered ash as the "mineral-rich powdery residue remaining on-site after a fire". However, there is some controversy regarding the ash definition between different disciplines in which ash is an element of study, such as geology, sedimentology, ecology or even industries that produce ash as a residue[2]. In the present thesis, the ash definition considered comes from the forest-fire research community. For instance, Raison in his review in 1979 (Raison, 1979) defined ash as "the material not entirely mineral in nature, deposited following burning of vegetation of its residues". Along these lines, Quill et al. (2010), defined as a "category of thermally altered plant species that have been submitted to intense temperatures in the presence of oxygen; whitish residues and relatively greater inorganic mineral content are the resultant physical and chemical properties" and distinguished ash from chars which "represent a continuum of thermally altered plant material resulting from relatively low burning temperatures, appearing brown to black and exhibiting greater carbon fractions than ash". Between Raison (1979) and Quill et al. (2012), others also defined ash similarly: Giovannini (1994), Forbes et al. (2006), Kinner and Moody (2010) or Scott (2010). However, this definition generates three main problems in relation to attributing a fire product the category of "ash": (i) the minimum temperature reached, (ii) the maximum size of the

[2] References related to ash from wood industry are not considered in any of the parts of the review or the thesis, unless indicated. Despite some similarities between ash from the wood industry and ash from wildfires it is considered here that the combustion conditions and temperatures and not sufficiently comparable. Even ash derived from different wood industries (pulp mills, woods products, wood as a fuel) is not directly comparable (Someshwar, 1996; Agusto, 2008). Furthermore, the effects of the application of ash on the forest soil and vegetation are not of the same nature as the effects of the ash deposited after wildfire, since the ash addition is manual and largely homogeneous while after fire, ash is patchily distributed, derived from mixed species and burn severities and interacts with a soil affected by fire. However, ash properties from wood are considered when material has been burned in a muffle furnace under laboratory conditions that are similar to the laboratory conditions used in wildfire research (e.g. Etiegni and Campbell, 1991; Someshwar, 1996; Demeyer, et al., 2001).

particles and (iii) the quantities of organic and inorganic material. Regarding the minimum temperature reached, it is considered that until 200 ºC only water evaporation takes place and oxidation does not occur. Therefore, oxidation must be take place in order to consider the material as ash (Grier, 1975; Misra et al., 1993; Úbeda et al., 2009; Quill et al., 2010). Concerning its maximum particle size there are different views. Wanthongchai et al. (2008) sieved ash to 0.5 mm, Moreno and Oechel (1991) 1 mm and Stoof et al. (2010) to 2 mm, with char or charcoal considered to be the bigger organic particles. The organic carbon content range of ash is not well established and from the few reports in the literature, it can vary from 0 to almost 100% (Table 1).

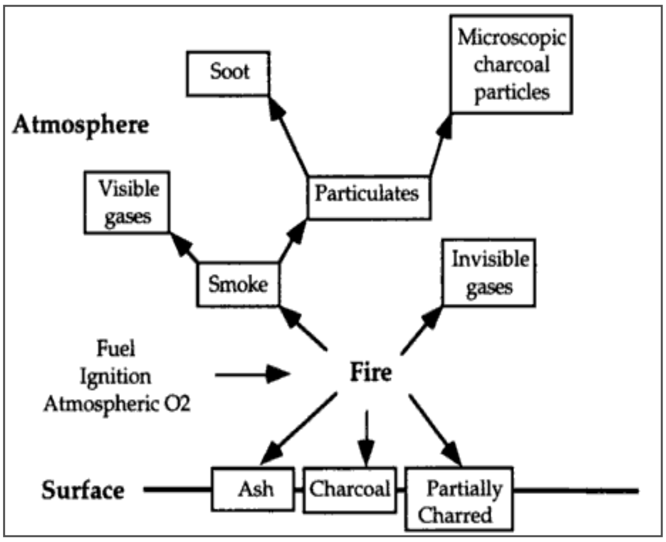

Figure 6. Relationships and terminology of fire products according to Jones et al. (1997).

The proportions of the type fire products yielded and their characteristics depend on the properties of the organic material burned (arrangement, size, density, moisture content and chemical composition) and on the type of the combustion and temperatures under which they are produced. If the organic material is only pyrolysed by heat before the ignition, charred material is mostly formed. During the flaming phase, temperatures can be between 300-1400 ºC and fire can spread rapidly, yielding a mixture of charred particles and ash whereas during the smouldering phase where the combustion stays in the same fuel until consumed and can be flameless, oxidation is more intense or even complete, producing very white ash and greater particulate emissions (DeBano et al., 1998). In this way, fire residues remaining in the soil and its properties (which will be considered in the next section 1.3.2.) can indicate the nature of the combustion and hence give information about nature of the fire itself. For that reason, ash has been used as an indicator of fire severity (Smith and Hudak, 2005; Lentile et al., 2009; Roy et al., 2010) usually in conjunction with other indicators such fuel consumption (e.g. thickness of the remaining branches), plant mortality, and the characteristics of the litter layer, roots or soil water repellency (Moreno and Oechel, 1989; Keeley, 2009; Parsons et al., 2010).

Table 1. Total organic carbon content (TOC, %) of ash reported in the literature.

Ash type	TOC (%)	Method	Reference
Bushfire, northeast Victoria (Australia)	21	Loss on ignition (800 ºC 18 h)	Bennet et al. (2004)
Pinus contorta, Pinus ponderosa and *Pseudotsuga* menziesii wood, burned in a barrel	47 - 66	Loss on ignition (400 ºC, 15h)	Burns (2007)
Wildfire, *Adenostoma fascilacutum*, California	38	Loss on ignition (700 ºC)	Christensen, 1973
Wildfire, northern territory (Australia), 26 samples	11 - 44	-	Forbes et al. (2006)
Clearing burn, tropical rainforest	19 - 33	Dry combustion in a high temperature induction furnace	Kauffman et al. (1995)
Burning of pastures, Amazon basin	16 - 20	Dry combustion in a high temperature induction furnace	Kauffman et al. (1998)
Eucalyptus pauciflora litter, burned in an oven	34	Dry combustion in a high temperature induction furnace	Khanna et al. (1994)
High severity wildfire of Ponderosa Pine forest	14.7	Loss on ignition	Larsen et al. (2009)
Eucalyptus tricarpa bark and leaves burned in an oven at 300 ºC	40 - 41	TOC Analyser	Quill et al. (2010)
Eucalyptus tricarpa bark and leaves burned in an oven 400 ºC	0 - 0.2		
Burned in the open with large metal trays: *Bothriochloa ambigua* S. T. and *Danthonia spp* (Straw from native pasture) Wheat straw	14 2.5	-	Raison and McGarity (1980a)
Low intensity prescribed burn in *Eucalyptus pauciflora* forest: Wood, grey ash Large twig, black ash Small twig, black ash Small twig, grey ash Thick bark, grey ash Thin bark, black ash Thin bark, grey ash Tree leave, black ash Shrub leave, black ash	 17.3 98.6 97.3 55.0 11.7 95.6 14.9 93.7 95.6	Loss on ignition	Raison et al. (1985)
Extreme severe bushfire in Victoria (Australia), eucalyptus forest of mixed species (12 samples)	6.0 - 8.5	TOC = TC-TIC TC = dry combustion at 1050°C. TIC = dry combustion at 1050°C, with a prior heating 4 h at 450°C to eliminate the organic C	Santín et al. (2012)
Moderate to high severity bushfire in Victoria (Australia), temperate rainforest (5 samples)	11.8 - 18.6		
Controlled fire (fuel up to 5cm diameter): *Pseudotsuga menziesii,* Pinus ponderosa and *Larix occidentalis* *Pinus contorta*	28.2 43.4	Loss on ignition (Nelsons and Sommers, 1996)	Woods and Balfour (2010)

1.3.2. Ash physical, chemical and mineralogical properties

In other to characterise ash, many analysis have been carried of ash collected from wildfires and generated in experimental burns (e.g. Grier, 1975; Marion et al., 1991; Ulery et al., 1993; Goforth et al., 2005; Marcos et al., 2009; Woods and Balfour, 2010; Pereira et al., 2012) or using furnaces in the laboratory (e.g. Etiegni and Campbell, 1991; Misra et al., 1993; Liodakis et al., 2005; Úbeda et al., 2008; Quill et al., 2010). Although conditions of the furnace are not exactly the same as in a wildfire, muffle furnaces are useful to control the temperature of production, which it is not possible in a wildfire. However, muffle experiments carried out do not follow an standardised methodology to burn the vegetation and have used different heating durations, initial moisture content, size of the crucibles and size of the fuels (e.g. White et al., 1972; Raison, 1979; Misra et al., 1993; Soto and Diaz-Fierros, 1993; Gabet and Bookter, 2011). These variables will affect the ignition, combustion and maximum temperature reached in the furnace. Procedures to standardise the methodology for the ash created under laboratory conditions are suggested in Pereira et al. (2010).

Once the ash is obtained, from either a wildfire, prescribed fire, experimental fire or laboratory experiments, the methods used to analyse ash properties are largely the same as used for soil, although some methods are more appropriate than others considering the nature of ash. For instance, the low bulk density of ash requires adding less sample quantity or more water for the pH measurement, or the thinness of the ash layer makes it difficult to measure hydraulic conductivity in the field. A description of the various methods used for ash analysis is given in Pereira et al. (2010). Most studies examining ash properties have focused on its chemical composition. During the 1940-1960s this was a popular topic and was related to the study of the forest nutrient cycle, as demonstrated in the reviews of Ahlgren and Ahlgren (1960) and Raison (1979). Mineralogical composition was studied especially since Etiegni and Campbell (1991) and only recently, physical and hydrological properties of ash have been given more attention. However, it is difficult to give a relevant standard characterisation of ash. Being a product of fire, ash properties are very variable, reflecting the combustion conditions, the type of vegetation and part of the plant burned (wood, leaves or bark), climate and soil type (Etiegni and Campbell, 1991; Marion et al., 1991; Ulery et al., 1993; Khanna et al., 1994; Someshwar, 1996; Gabet and Bookter, 2011). As an example, Úbeda et al. (2009) found differences in mass loss, colour, electrical conductivity and Ca:Mg ratio after burning at the same temperatures the leaves of the same tree specie (*Quercus suber*), but obtained from different sites in central Portugal and northeast Spain. They attributed these differences to the different flammability of the leaves due to differences in leave shape, with the Portuguese ones allowing better oxygen circulation and therefore more intense thermal degradation. Ash properties are also very changeable in time with the interaction of CO_2 from the atmosphere and especially with rain water (Etiegni and Campbell, 1991; Balfour and Woods, 2006).

Despite this variability in ash properties, the next section attempts to classify ash properties according to a range of combustion temperatures, but it has to be kept in mind that temperatures indicated and properties may vary depending on the vegetation considered. The variability of the ash properties is one of the main factors explaining the different effects of ash on ecosystem components (section 1.3.3).

Chemical and mineralogical properties

From 150-200 ºC, once thermal alteration started, and approximately until 350 ºC, the resultant ash is a mixture of biomolecules thermally altered during an incomplete combustion (small particles of char and charcoal) and a small proportion of particles already mineralised (Baldock and Smernik, 2002; Bennet et al., 2004; Forbes et al., 2006). The organic carbon of this ash is often referred also as black carbon (BC) and is rich in organic compounds of aromatic and carboxylic character of low solubility (BC; Almendros et al., 1992; Forbes et al., 2006; Quill et al., 2010; Scott, 2010; Dlapa et al., 2012). Despite that, there are recent studies on the significant quantities of dissolved organic carbon (DOM) from ash and contains small-sized polyaromatic hydrocarbons, which are well known pollutants in waters (Smith et al., 2011; Santín, et al., 2012). This DOM is almost non-existent in samples heated above 400 ºC and the quantity and nature of the extracted DOM varies with the species from which residues were derived. For instance DOM derived from shrubs were higher than for herbaceous species (Quill et al., 2010; Zhao et al., 2010). The mineral components of this ash are low, resulting in a pH and electrical conductivity very similar to the vegetation without burning (Goforth et al., 2005; Úbeda et al., 2009).

As temperature increases from 350 to 450-500 ºC, oxidation is more intense and organic compounds are reduced, but still existent (Table 1; Balfour and Woods, 2010; Quill et al., 2010). Ash colour becomes lighter (White et al., 1972; Pereira et al., 2012), pH increases by two to three units and electrical conductivity also increases due to the major presence of inorganic elements. These are mainly, and depending on the specie, calcium (Ca), magnesium (Mg), potassium (K), silicium (Si), and in lower proportion phosphorous (P), sodium (Na), sulphur (S), other metals such as aluminium (Al), iron (Fe), manganese (Mn) and zinc (Zn) (Etiegni and Campbell, 1991; Johansen et al., 2003; Goforth et al., 2005; Wanthongchai et al., 2008; Qian et al., 2009; Úbeda et al., 2009; Gabet and Bookter, 2011). Some of these elements increase their relative proportions in ash as the temperature of combustion increases and the amount of others decrease due to their lower temperatures of volatilisation. N and S can volatilise at 200-300 ºC, P around 550-750 ºC and all the other elements need higher than 750 ºC to volatilise (Raison et al., 1985; Weast, 1989; DeBano et al., 1998). In these range of temperatures, the elements on ash are present mainly as carbonates, bicarbonates, chlorides, nitrates, sulphates and phosphates, although the relative amounts are species-dependent (Liodakis et al., 2005; Quill et al., 2010).

Above 450-500 ºC, provided sufficient oxygen is available, an almost complete combustion of the organic compounds occurs leaving a light grey or white coloured-ash of high pH, reported to range from pH 8 to 12 (Etiegni and Campbell, 1991; Ulery et al., 1993; Goforth et al., 2005; Marcos et al. 2009; Qian et al., 2009; Úbeda et al., 2009). At these high temperatures, the ash from different species becomes more chemically similar as the combustion temperature increases and the main differences with temperature are the proportions of carbonates and oxides. Around 500-700 ºC the main minerals are carbonates and above the 1000 ºC the basic cations are transformed into oxides (Etiegni and Campbell, 1991; Goforth et al., 2005; Quill et al., 2010; Pereira et al., 2012). From the carbonates, calcium carbonate or calcite ($CaCO_3$) is particlualy abundant, followed by magnesium and potassium carbonate ($MgCO_3$, K_2CO_3; Demeyer et al., 2001). The most common oxides and hydroxides are calcium oxide (lime; CaO), magnesium oxide (MgO), quartz (SiO_2) and portlandite ($Ca(OH)_2$). Other minerals such as as calcium silicate (Ca_2SiO_4) or calcium and potassium sulphates (K_2SO_4) are also commonly present (Etiegni and Campbell, 1991; Misra et al., 1993; Goforth et al., 2005; Liodakis et al., 2005; Balfour and Woods, 2010; Quill et al., 2010; Pereira, 2011).

It must be borne in mind that new carbonates, bicarbonates and oxides can be formed as soon as the ash is deposited at the surface and interacts with atmospheric CO_2 or with rainwater (Demeyer et al., 2001). Using the calcium salts as an example (Etiegni and Campbell, 1991; Balfour and Woods, 2006), the calcium oxide can be hydrated and form portlandite:

$$CaO + H_2O \Rightarrow Ca\ (OH)_2$$

And carbonation reactions can occur in portlandite and calcium carbonate resulting in the formation of calcium carbonate and water in the former and bicarbonates in the latter:

$$Ca\ (OH)_2 + CO_2 \Rightarrow CaCO_3 + H_2O$$

$$CaCO_3 + H_2O \Rightarrow Ca\ (HCO_3)$$

Physical properties

Mass loss with respect to the biomass before the combustion on ash produced at temperatures lower than 350 ºC can be 50-60% due to the mineralisation and volatilisation of some of the elements that compose ash (Úbeda et al., 2009). Regarding its particle size, Woods and Balfour (2011) burned in the laboratory vegetation of *Pinus contorta, Pinus ponderosa* and *Pseudotsuga* menziesii at different temperatures and the ash produced at 300 ºC was the coarsest (average of the median diameter D_{50} = 90 µm) compared with other temperatures (Figure 7). This ash also had the highest saturated hydraulic conductivity, K_{sat} = 3600 mm h^{-1}, reflecting the dominance of charred material in the matrix (Balfour and Woods, 2010). The density of this type of ash type with abundant charred material is low with particles able to even float in the water (Mulleneers et al., 1999; Rumpel et al., 2006).

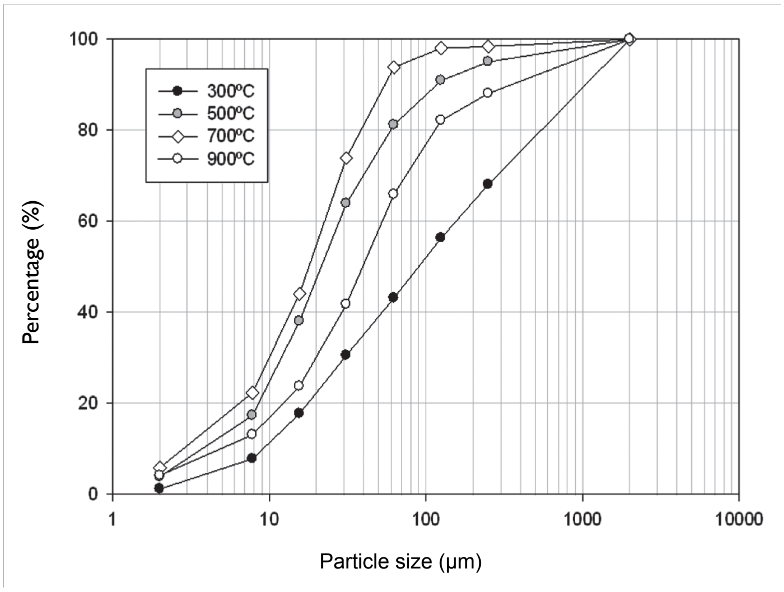

Figure 7. Particle size distribution of ash produced in the laboratory at 300, 500, 700 and 900 ºC from *Pinus contorta, Pinus ponderosa* and *Pseudotsuga* menziesii. Particle size decreases in the 300-700 ºC range and then increases for the 900 ºC samples. From Woods and Balfour (2011).

At temperatures higher than 450-500 ºC, around 80% of the biomass is lost according to laboratory experiments (Etiegni and Campbell, 1991; Úbeda et al., 2009). The particle size of the vegetation burned by Woods and Balfour (2011) at 500 ºC and 700 ºC (Figure 7) became finer as the temperature increased (D_{50} 500 ºC = 20 µm and D_{50} 700 ºC = 18 µm). The finer texture was thought to be associated with a more complete combustion and a transition from primarily organic fragments in the ash to primarily crystalline fragments of calcite and other minerals. However, at 900 ºC Woods and Balfour (2011) detected a further increase of particle size (D_{50}900 = 40 µm) associated with re-crystallization after the CaO hydration. Consistently with the coarsening of the ash texture, the saturated hydraulic conductivity of these samples showed a reduction at 500 ºC and 700 ºC and an increase at 900 ºC (K_{sat} 500 ºC = 126 mm h^{-1}, K_{sat} 700 ºC = 90 mm h^{-1}, K_{sat} 900 ºC = 720 mm h^{-1}). For ash from the same species used by Woods and Balfour (2011) and burned in a barrel, Bookter (2006) reported median particle diameters (D_{50}) between 30 and 120 µm. For *Pinus contorta* ash produced at high temperatures, Etiegni and Campbell (1991), reported an average diameter of 230 µm, which unfortunately may not be comparable with the median diameter stated above. As observed under a microscope, they described ash to have large and porous carbon particles and many irregular shaped inorganic particles.

Ash produced under field conditions exhibits more variable properties. In a moderate severity forest fire in Colorado, Kinner and Moody (2007) measured that the particle size of the ash and found that in north-facing plots it was finer (D_{50} = 750 µm) than in the south-facing plots (D_{50} = 1400 µm). The reasons they suggested are the type of species, having larger needles and branches in the south aspect that resulted in coarser ash (Figure 8). Different plant species besides the difference in temperature of combustion explained also the wide range of median particle size of 6 samples from two different wildfires in Spain, from D_{50} = 17 to 134 µm (Woods and Balfour, 2011). In an experimental fire in Montana, Woods and Balfour (2010) measured particle sizes of D_{50} = 147 and D_{50} = 244 µm. Regarding particle densities, values between 2.05 and 2.52 g cm^{-3} have been reported (Bookter, 2006; Cerdà and Doerr, 2008; Woods and Balfour, 2011). According to Kinner and Moody (2007) the particle density of the ash varied depending on the particle size, increasing in density as the particles are smaller. They results were: 1.7 g cm^{-3} in particles from 500 to 250 µm, 1.8 g cm^{-3} in particle form 250 to 63 µm and 2.4 g cm^{-3} for sizes smaller than 63 µm (the samples are the same as in Figure 8).

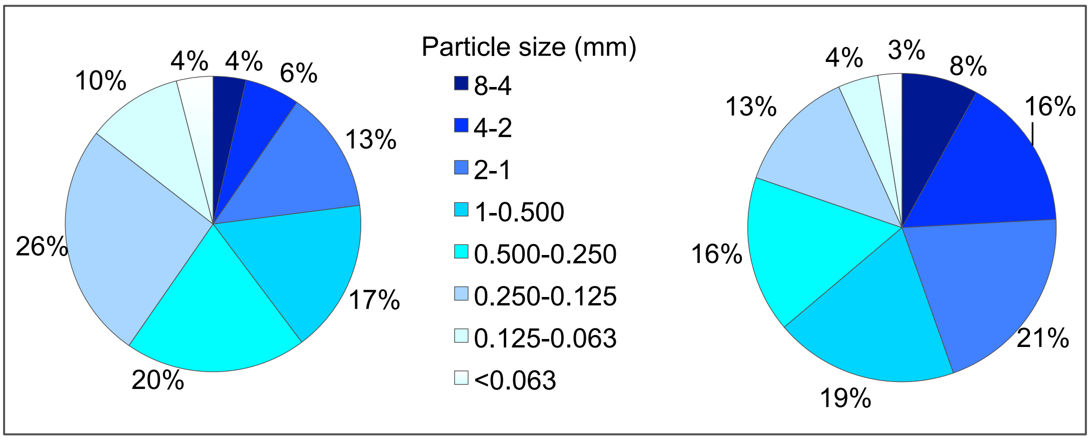

Figure 8. Particle size distribution (%) of ash collected in a hillslope. Left: North aspect with *Populus tremuloides* and *Pseudotsuga menziesii*. Right: South aspect with *Pinus ponderosa*. The latter had larger needles and branches, which resulted in coarser ash. After Kinner and Moody (2007).

Bulk density of ash is difficult to determine in the field due to the thinness and changeability of the ash layer with time. However, from the few measurements carried out on white ash, reported ash bulk densities range from 0.18-0.62 g cm^{-3} being higher as the layer is thicker (Goforth et al., 2005; Bookter, 2006; Cerdà and Doerr, 2008; Woods and Balfour, 2008; Moody et al., 2009). However, Massman et al. (2008) measured the highest bulk density of 0.96 g cm^{-3} in a prescribed experimental slash pile burn and did not found statistically significant differences with ash depth. In any case, the values from ash are in some cases considerably lower than the typical range of values for mineral soils. This reflects the high porosity of ash of around 60-80% and its great capacity to store water. It means that a layer of ash can store a depth of water approximately equal to half of its thickness before the initiation of runoff (Cerdà and Doerr, 2008; Woods and Balfour, 2008; Woods and Balfour, 2010; Gabet and Bookter, 2011). The saturated hydraulic conductivity in the field is also difficult to measure and the available data derives mainly from laboratory experiments with repacked ash. A wide range of values have been measured: K_{sat}= 21 to 65 mm h^{-1} in 6 samples from two wildfires in Spain (Woods and Balfour, 2011), K_{sat} = 56 mm h^{-1} in ash from an intense *Pinus ponderosa* wildfire (Woods and Balfour, 2008), K_{sat} = 600 and K_{sat} = 380 mm h^{-1} for the same vegetation, but produced in an experimental fire (Woods and Balfour,

2010) and K_{sat} = 165 mm h^{-1} in laboratory-produced ash (Gabet and Bookter, 2011). According to Woods and Balfour (2011) saturated hydraulic conductivity in ash is in the range of the expected for mineral soils with the same texture. These conductivities and water storage capacities indicate a material of hydrophilic nature (Etiegni and Campbell, 1991; Leighton-Boyce et al., 2007; Woods and Balfour, 2008), which has been confirmed only in few occasions with water repellency measurements (Cerdà and Doerr, 2008). However, ash have been reported few times of being difficult to wet from Californian chaparral (Gabet and Sternberg, 2008), *Eucalyptus sp.* forest in Australia (Khanna et al., 1996) and *Pseudotsuga menziesii* forest in Montana (Stark, 1977).

The physical properties of ash can also change with time as ash remains on the ground, especially after a rainfall event. Swelling of ash of 2% has been detected immediately after burning by Stoof et al. (2010) and of 12.5% after 4 weeks under water for ash produced at more than 500 ºC by Etiegni and Campbell (1991). They suggested that this might be due to the hydration of oxides. Balfour and Woods (2006) also observed a mineralogical transformation from calcium oxide (CaO) to portlandite ($Ca(OH)_2$) and calcite ($CaCO_3$), that resulted in an irreversible hardening and crusting for ash generated at 900 ºC compared to that produced at 100 or 500 ºC. This process is also known for industrial wood ash produced at high temperatures and has been termed by Steenari and Lindqvist (1997) "self-hardening". The resultant hardened ash has a substantially lower hydraulic conductivity and infiltration capacity than unhardened ash (Cerdà, 1998a; Balfour and Woods, 2006; Onda et al., 2008).

1.3.3. Ash effects on ecosystems components

After a wildfire, ash is deposited in the ground until it is removed off site by rainfall or wind, or incorporated into the soil by physical, chemical or biological processes. Raison (1979) indicated that after a fire the soil is directly affected by inputs of heat and ash and by a microclimate modification due to the removal of the vegetation. The effect of these three factors are difficult to separate although attempts have been made to examine the effects of heat and ash separately as well as its interactions (e.g. Raison and McGarity, 1980a, b; Khanna et al., 1996; Badia and Martí, 2003). The effects of heat on the soil have been reported in the previous sections (1.2.1., 1.2.2.). In this section the effects of ash on different parts of the ecosystem are examined as regards the physical and chemical properties of the soil, the forest nutrient cycle, the carbon cycle, microbial activity, plant growth, water quality and hydrological processes (Figure 9). The reader is reminded here that studies on ash effects are limited, although more data is available for the forest nutrient cycle.

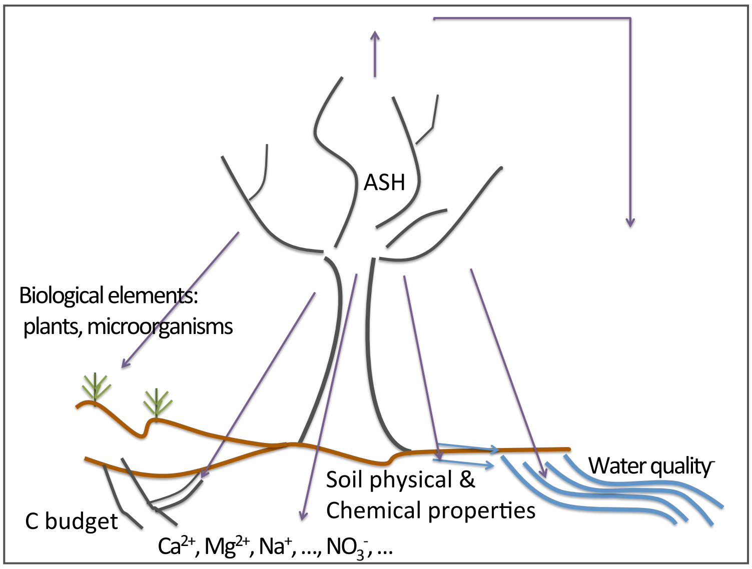

Figure 9. Ash can have an impact on different elements of the ecosystem: soil physical and chemical properties, the nutrient cycle, the carbon cycle, plants and microorganisms, soil erosion and runoff, and water quality.

Ash effects are not the same after every wildfire and not even within the same fire. The reasons are not only the variations in ash properties, which depend on the vegetation type and the temperature of combustion, but other factors as quantity of ash deposited on the ground, its spatial distribution, the time it remains on the soil and whether or not it is incorporated into the soil (DeBano et al., 1998; Raison et al., 2009). The thickness of the ash layer covering the ground can vary from small discontinuous quantities of dark residues in low severity fires to thick layers of white ash (Keeley and Zedler, 2009). Reported ash thicknesses range from 10 to 34 mm in a *Pseudotsuga menziesii* and scattered *Larix occidentalis* forest fire in Montana (Woods and Balfour, 2008); 24 to 48 mm in the Valencia province in Spain after a *Pinus halepensis* forest fire (Cerdà and Doerr, 2008); 50 mm in New Mexico in a mixed conifer forest consisting of *Pinus ponderosa* and *Pseudotsuga menziesii* (Cannon et al., 2001); 60 to 80 mm in a mixed conifer forest fire (*Abies concolor, Calocedrus decurrens, Pinus lambertioana, Pinus jeffreyi, Pinus ponderosa and Quercus chrysolepsis*) in California (Goforth et al., 2005); 12 mm after a fire of *Pinus ponderosa* in Colorado (resulting in 6.3 kg m^{-2} of ash; Larsen et al., 2009), and from 10 to 38 mm in mixed species eucalypt fire in Victoria, Australia (8.2 kg m^{-2}; Santín et al., 2012). However, the ash does not cover the soil homogeneously. A complete (100%) ash cover has been reported by Cerdà and Doerr (2008), but in similar Mediterranean ecosystems Lavee et al. (1995) reported a 30% cover and De Luis et al. (2003) 30.9%, 44.5% and 63.8% in different plots where ash cover increased with fire severity. After a high severity fire in a *Pinus ponderosa* forest in Colorado, Larsen et al. (2009) measured an initial cover of 54%.

The non-homogeneous effects of fire are reflected also in the variability of colour of ash on the ground that usually varies between patches of white, grey and black ash. For instance, Marcos et al. (1999) reported 67.5% of white ash and 32.4% of black ash produced in a plot of *Pinus radiata* forest, while a slash and burn plot had 86% of white ash and 4% of black ash. The slash logs can combust and smoulder for a long time leaving thick deposits of white ash, termed "ash bed" by Australian scientists (Humphreys and Lambert, 1965; Chambers and Attiwill, 1994). In the field, white ash usually is deposited on top of a layer of black ash, as the leaves from vegetation are combusted at higher temperatures with more oxygen available and deposited over the charred litter layer (Lewis, 1974; Blank and Zamudio, 1998).

Once ash is deposited, its effects depends on the varied time it remains on the ground. Mataix-Solera (1999) reported that one month after an experimental fire in the Sierra de Aitana (Province of Alicante, Spain) the ash cover was almost completely removed after only one day with wind speeds reaching 90 km h^{-1}. After a year, ash is usually completely removed or mixed with sediments or the soil particles (González et al., 1991; Blank and Zamudio, 1998; Larsen et al. 2009). Although the fate of ash has not been studied much, it can be expected to be incorporated into the soil, deposited in footslopes or depressions or joining the fluvial system and forming part of rivers, lakes and, ultimately, marine deposits (Figure 10; Cerdà, 1998a; Schmidt and Noack, 2000; Shin et al., 2002; Novara et al., 2011; Santín et al., 2012).

Figure 10. Left: Ash and sediments accumulated on a lower slope in the Sierra de Enguera after an storm of 15 mm in 10 mins two and a half month after the fire. Right: Ash layer in the profile of a colluvial soil in the Serra Grossa, Vallada, Valencia, Spain (Photograph taken by Artemi Cerdà).

1.3.3.1. Ash effects on soil properties and the nutrient cycle

Nutrients in a forest are stored in the rocks, litter layer, soil or in the living organisms and can be transferred between these compartments along a variety of pathways such as weathering, decomposition, mineralisation or through the food chain. The rate at which nutrients move in this cycle depends in general on the climate and living organisms, but it can be different at micro-scale depending on the aspect or geomorphological location (Figure 11 left; Attiwill and Leeper, 1990; DeBano et al., 1998). After a fire, the substances that would otherwise be locked into the nutrient reservoir of the litter and living vegetation are released much faster that would occur from biological decomposition. Some nutrients may be rendered volatile, but many others will be released onto the ground in the form of charcoal and ash (Figure 11 right; DeBano and Conrad, 1978; Lewis, 1974; Raison et al., 2009). Some of the ash may be available for incorporation into the soil and uptake by plants, but some might be redistributed horizontally by wind or surface erosion or downwards by leaching into the soil (Lewis, 1974; Boerner, 1982; Khanna et al., 1994; Cerdà and Doerr, 2008). The ash deposition and incorporation into the soil also have some effects on soil physical properties.

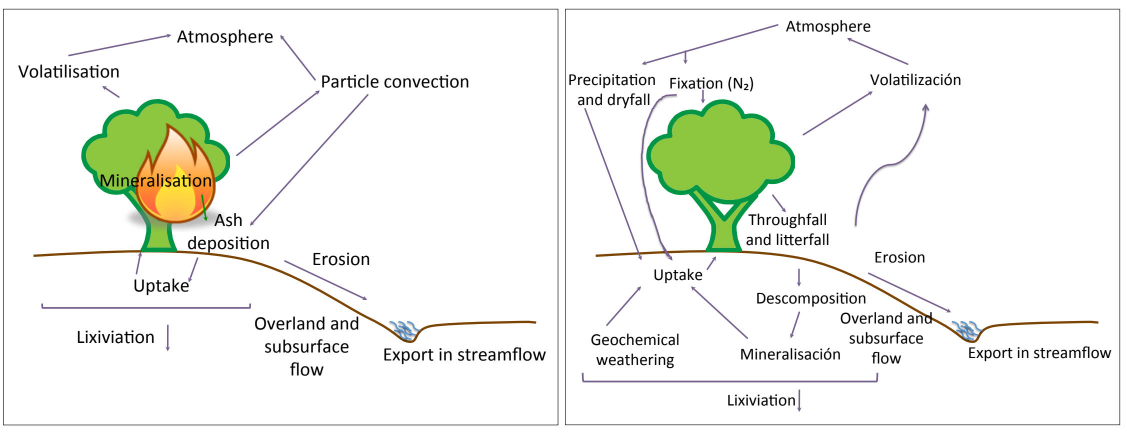

Figure 11. Left: Nutrient cycle in natural environments. After Brown (1980). Right: Nutrient cycling after a forest fire.

Effects of ash on soil chemistry

When ash is produced at temperatures lower than 500 ºC, a considerable amount of charred organic carbon and black carbon might be incorporated into the soil by biological or physical mechanisms and become part of the soil organic matter (Czimczik and Masiello, 2007; Eckmeier et al., 2007). Once the pyrogenic organic carbon is incorporated into the deeper mineral soil, it is protected to some degree from further oxidation and loses off site by erosion (Rumpel et al., 2009). As a consequence, the soil organic matter may change its quality, for example, increasing its aromaticity (Knicker, 2007). Elements such as chloride (Cl^-), phosphorous (P) and ammonia (NH_4^-) have also been found to increase after low severity fires due to ash incorporation into the soil (Christensen, 1973; Raison, 1979; Khanna and Raison, 1986; Marion et al., 1991; Khanna et al., 1994; Qian et al., 2009).

The incorporation of ash produced at temperatures much above 500 ºC is likely to affect more strongly the inorganic component of the soil. The mineral and alkaline white ash is also more soluble than the black ash (Raison and McGarity, 1980a; Úbeda et al., 2009). For instance, after carrying out different leaching experiments Bennet et al. (2004) reported that the solubility of the charcoal was approximately 100 times lower than the white ash. White ash may increase soil pH by up to 3 units, depending on the initial soil pH (Raison and McGarity, 1980b; Kutiel and Naveh, 1987b; Thomas and Wein, 1990; Soto and Diaz-Fierros, 1993; Andreu et al., 1996; Khanna et al., 1996; Blank and Zamudio, 1998; Badia and Martí, 2003; Alauziz et al., 2004; Molina et al., 2007) and with the magnitude of change being dependent on the type and quantity of ash added, and the soil buffering capacity (Raison and McGarity, 1980b; González et al., 1991; Soto et al., 1997; Molina et al., 2007; Marcos et al., 2009). After ash deposition, substantial increases in increases in Ca^{2+}, Mg^{2+} and K^{+} have also been reported (Humphreys and Lambert, 1965; White et al., 1972; Grier, 1975; Khanna and Raison, 1986; Kutiel and Naveh, 1987b; Dyrness et al., 1989; Thomas and Wein, 1990; Marion et al., 1991; Carreira and Niell, 1995; Andreu et al., 1996; Khanna et al., 1996; Ludwig et al., 1998; Badia and Martí, 2003; Alauziz et al., 2004; Johnson et al., 2005) and in lower quantities of Na, Al, Fe, Mn, Zn and Si (Kutiel and Naveh, 1987b; Thomas and Wein, 1990; Khanna et al., 1994; Qian et al., 2009; Pereira and Úbeda, 2010). These increases have been detected particularly in the first 2.5-5 cm of soil (Marion et al., 1991; Ludwig et al., 1998; Molina et al., 2007).

Depending on the solubility of the ash and the type of soil, however, only some of the total ash incorporated into the soil will become a part of the soil solution and thus available for plant uptake. The solubilisation of the nutrients in ash depends on: (i) the mineral components of ash, as oxides and bicarbonates are more easy to dissolve than carbonates, which have been found to still be present three years after fire (Soto and Diaz-Fierros, 1993; Ulery et al., 1993; Steenari and Lindqvist, 1997), and on (ii) the availability of water and moisture conditions after the fire, with the first substantial storm releasing most of the nutrients (Lewis, 1974; Grier, 1975; Stark, 1977; Soto et al., 1997; Lasanta and Cerdà, 2005). Khanna et al. (1994) grouped the elements into three categories according to their solubility: (a) most of the K, S and B (boron) were water soluble in part, whilst about 30% of the amount remained insoluble; (b) much Ca, Mg, Si and Fe dissolved progressively as more water was added to the ash; (c) P was mostly insoluble in water and required protons for dissolution, making it more available in acidic soils. Once nutrients are dissolved their permanence depends on the type of soil, particularly its pH, cation exchange capacity (CEC) and microbial activity (Humphreys and Lambert, 1965; Marion et al., 1991; Khanna et al., 1994; Khanna et al., 1996; Ketterings and Bighman, 2000; Demeyer et al., 2001).

According to the rate of ash nutrient incorporation into the soil and the quantity of ash lost, the increase in pH and the availability of nutrients can be long-, medium- or short-term. Johnson et al. (2005) found greater exchangeable K^{+}, Ca^{2+}, and Mg^{2+} attributed to the ash effect after as much as 20 years. Úbeda et al. (2005), reported higher pH values after 7 years in burnt plots compared to unburned ones and Humphreys and Lambert (1965) 9 years after fire under where it used to be an ash bed. In contrast, shorter-term effects are due to ash losses after a fire. Soil chemical changes during the first year have been detected by Kutiel and Naveh (1987a, b), Soto et al. (1997) and Marcos et al. (1999); during the first 4 months by Mataix-Solera (1999) due to ash removal by wind in the first few months after the fire; and within only 12 weeks by Ketterings and Bighman (2000) in a slash and burn fire.

The ash losses off-site contribute to a reduction of the total nutrients in the ecosystem (Neary et al., 1999; Raison et al., 2009). Losses by wind or water erosion are important in the context of ash and fire because ash is a very erodible material and burned terrain is particularly exposed to the agents of erosion due to the loss of protective vegetation and litter cover. These losses include soluble and non-soluble nutrients and also pyrogenic carbon (DeBano and Conrad, 1978; Thomas et al., 1999; Rumpel et al., 2009). The nutrient losses will depend on the fire severity, the steepness of the terrain and the weather (wind and rainfall) during and immediately following the fire, since most of the losses occur the first year after the fire (Carreira and Niell, 1995; Andreu et al., 1996; Gimeno-García et al., 2000; Ranalli, 2004; Lasanta and Cerdà, 2005). Losses by erosion given in DeBano and Conrad (1976), in a prescribed burned chaparral watershed were: 15.08 kg ha^{-1} of N, 3.37 kg ha^{-1} of P, 19.34 kg ha^{-1} of K, 28.02 kg ha^{-1} of Mg, 47.39 kg ha^{-1} of Ca y 2.57 kg ha^{-1} of Na. Other losses by erosion have been reported to be of 3-4 orders of magnitude in an eucalypt forest in Portugal during the during the first three years after the fire (Thomas et al., 1999) and of one order of magnitude in a gorse scrubland in the South of Spain two months after a prescribed fire (Carreira and Niell, 1995).

Losses by lixiviation are not too large compared with erosion, except in coarse soils that have little cation exchange capacity and after very intense fires (Khanna and Raison, 1986; Raison et al., 2009). Marcos et al. (1999) also found that all the nutrient losses after a fire in a *Pinus radiata* forest were lost due to lixiviation, as well as Lewis (1974) in a South Carolina pine forest and Smith (1970) in a *Pinus banksiana* forest. DeBano and Conrad (1976) quantified the losses by lixiviation in a prescribed burned chaparral watershed to be 7.67 kg ha^{-1} of K, 3.63 kg ha^{-1} of Mg, 20.04 kg ha^{-1} of Ca and 2 kg ha^{-1} of Na.

Effects of ash on soil physical properties

After the addition of ash produced at low fire severity, a darkening in soil colour has been observed (Badia and Martí, 2003). This darkening may affect the thermal properties of the soil, decreasing its albedo. However, Massman et al. (2008), after measuring soil and ash thermal properties in a slash experimental burn, suggested that that the presence of an ash layer (of non reported colour) insulates the soil proportionally with the ash layer depth, thereby reducing the temperature extremes that would otherwise occur in the soil.

Soil texture has been reported to not change by Badia and Martí (2003) adding 1% and 0.5% of black ash by weight of soil (5 g kg and 10g kg^{-1}) and Stoof et al. (2010) adding 15.5% ash by weight (equivalent to 1 cm of ash in the soil). However, Durgin (1985) observed that ash leachate was able to disperse the soil particles, but only for a developed soil, which had more hydroxides and pH-dependent charge than weakly developed soils. Similar effects of ash on soil particle size were reported by Giovannini (1994) in the hours immediately following the ash leachates application to the soil, however, as the contact time increased with ash, soil and water, there was a rise in the aggregation of finer clay particles into silt-sized fraction. This aggregation was also reported by Holcomb and Durgin (1979) who sprayed a plot with severely burned white ash leachates. The flocculation of the clay size also reduced soil erodibility.

Ash can also modify the soil hydraulic properties. Stoof et al. (2010) quantified an increase in soil water retention due to the ash addition to the soil. Similar results were observed by Chang et al. (1976) and Campbell et al. (1983) after fly ash addition on the soil. However, ash is also reported to clog soil pores, reduce soil porosity and infiltration capacity (Mallik et al., 1984; Balfour and Woods, 2007). Therefore, the increase in soil water retention as hypothesized by Stoof et al. (2010) is suggested to lie in the swelling nature of ash. For a soil containing ash, the volume of water stored at saturation does not only account for water stored in pores (i.e. soil porosity), but also for the volume of water absorbed by the ash particles.

1.3.3.2. Biotic response to ash addition on the soil

It is widely accepted that ash stimulates microbial activity due to the chemical changes produced when it is incorporated in the soil. According to the reviews of Raison (1979) and Mataix-Solera et al. (2009) this may be the cause of the post-fire increase in the bacteria/fungi ratio, given that a high pH favours bacteria. However, the changes seem to vary with soil type. Raison and McGarity (1980a) reported increases in respiration rate in sandy podzolic soils with any dose of ash addition, but for the more highly buffered krasnozem soil, high rates of ash slightly decreased respiration rate, while small amounts had no effect. In addition, ash did not stimulate respiration rate in steam-sterilised podzolic soil, which indicates that ash exerts influence on already active soil biological populations.

Regarding the effects of ash on seed germination, several studies suggest that a thick layer of ash (more than 2 cm) has a negative effect on seed germination of a range of species including *Pinus Pinaster, Pinus radiata, Eucalyptus globulus* (González-Rabanal et al., 1994); *Avenula marginata*, *Calluna vulgaris, Erica umbrellata* (González-Rabanal and Casal, 1995), *Pinus halepensis, Cistus salvifolius* (Ne'eman et al., 1993b; Izhaki et al., 2000) and *Pinus Balksiana* (Thomas and Wein, 1990). However, ash solutions seem not to have any effect (Nee'man et al., 1993b; Escudero et al., 1997; Herrero et al., 2007). Hypotheses of the inhibition of germination by ash are that it may prevent water from reaching the seed given the high osmotic pressure in the medium at which some species are sensitive, or maybe the seed is poisoned by the toxic effects on certain ions (Ne'eman et al., 1993a) or by the alkaline pH of the solution (Thomas and Wein, 1990).

In contrast, regarding vegetation growth, experiments in greenhouses with different species (*Lactuca sativa*, Vlamis and Gowans, 1961; *Hordeum sativum* and *Medicago sativa,* Kutiel and Naveh, 1987b) have shown that plants are positively affected by the addition of ash nutrients such as Ca^{2+}, Mg^{2+}, K^{+}, P and N fixation stimulated by fire during one year. This is especially important under ash beds because fertilisation effects operate in the longer term, producing the 'ash bed effect' (Raison et al. 1979; Chambers and Attiwill, 1994; Raison et al., 2009).

1.3.3.3. Ash effects on runoff production and soil erosion

There have been few studies to date examining the effects of ash on overland and subsurface flow processes, and on soil erosion compared to those focussing on ash effects on soil chemistry and the nutrient cycle. One reason for this may be that very few studies have examined soil hydrological processes immediately after a fire, when ash still covers the soil and its effects may be apparent.

Until 2000, most of the reports on ash effecting soil hydrology were only qualitative observations. Stark (1977) described that during the rains after the forest fire, soil permeability was limited, among other factors, due to ash forming a greasy cover. Mallik et al. (1984) reported from a heathland in the N.E. Scotland, that soil covered with ash had lower infiltration rates, but that it unexpectedly increased the soil's water retention capacity. They deduced that ash with high water retention capacity clogged the pores in the upper soil layer and suggested that this should be explored with observations of soil thin sections. Also Lavee et al. (1995) in a *Pinus halepensis* forest in Israel as well as Martin and Moody (2001) in *Pinus ponderosa* forests of Colorado and New Mexico, pointed out that among other factors, the clogging of pores by ash could be the reason of increased overland flow. However, in the same paper, Martin and Moody reported that during the infiltration experiments ash acted as a storage reservoir for rainfall and prevented runoff. This effect was also reported by Cerdà (1993; 1995; 1998b) in *Pinus halepensis* forests in Spain, where the high infiltration capacity and water retention capacity of ash was one of the reasons why overland flow was not significantly increased immediately after the fire. Gimeno-García et al. (2007) in a Mediterranean shrubland area also attributed to ash the low soil erosion rates that prevented splash erosion.

Recent studies focused specifically on ash effects on soil hydrology. To achieve that, mostly rainfall simulations were used to compare runoff and erosion rates obtained from ash covered and control plots in a paired-plot strategy approach. Woods and Balfour (2008) demonstrated in Western Montana that ash can temporarily, but substantially, reduce post-fire runoff and erosion rates in the period immediately following severe wildfires (Table 2). They attributed the reduction and delay of runoff to the storage of water in the highly porous ash layer and to the fact that the ash protects the underlying mineral soil from surface sealing. The reduction in erosion from ash-covered soils was primarily due to this reduction in runoff caused by the ash layer, and to the rain splash detachment reduction by ash soil protection. The same effect of ash delaying overland flow by storing water and the reduction in sediment yields was reported immediately after a wildfire in an *Eucalyptus* forest in Portugal under wettable and water repellent conditions (Leighton-Boyce et al., 2002), in *Populus temuloides* and *Pinus ponderosa* wildfire in Colorado (Kinner and Moody, 2007), in a wildfire in *Pinus halepensis* forest in Spain (Cerdà and Doerr, 2008), in a heathland in the south of Spain, where hydrological response and erosional response after a one-year period were comparable to pre-fire conditions at a plot scale (Zavala et al., 2009), and in laboratory rainfall simulations using two different soils (granitic and micaceous) and ash collected in a *Pinus ponderosa* fire using two ash thickness (0.5 cm and 1.2 cm; Larsen et al., 2009).

As ash thickness increased, runoff coefficients, final runoff rates, and sediment yields decreased (Table 2; Larsen et al., 2009). They also observed thin sections of soil and observed no sign of pore clogging, as ash was coarser than soil pores. However, Woods and Balfour (2010), adding ash from similar species, noted that if the ash layer was thinner than 1 cm there was a reduction in the infiltration rates while layers of 2 cm and 5 cm delayed and reduced total overland flow. This phenomenon only occurred in the sandy loam soil because ash clogged the soil pores of this coarser soil one whereas the finer texture of the other soil prevented pore clogging, even after 10 months. Gabet and Sternberg (2008) also attributed to pore clogging the reduction of infiltrability in their coarse soil in flume experiment. Furthermore, Onda et al. (2008) detected increases in the runoff response by a factor of 4 relative to the first post-fire storm in a *Pinus Muricata* and *Pseudotsuga menziesii* forest in California. They attributed the increases to surface sealing to a low conductivity ash layer formed by the raindrop impacts that compacted and crusted the ash after the first storm. The same type of crust and a low conductivity layer was reported by Woods and Balfour (2008) 10 months after a fire in Montana.

The quantification of the effects of ash on runoff and the analysis of its physical and chemical properties are shedding light on the different roles of ash in the hydrological processes immediately after a wildfire. The main reasons of this variable response reported can be attributed to three main sets of variables.

1) The different properties of the soil and ash composing the two layer system:

(i) texture and particle sizes, either from soil or ash, control whether or not ash is able to clog soil pores (Balfour and Woods, 2007; Larsen et al., 2009; Woods and Balfour, 2010);

(ii) in cases where ash does not clog soil pores, the hydraulic conductivity of the ash (K_{ash}) and the soil (K_{soil}) determine whether ash or soil exerts the control in the infiltration process. If $K_{ash} > K_{soil}$, the lower layer (soil) controls infiltration and water ponds from the interface to the surface. Ash acts as a capillary barrier and thus stores water rather than controlling infiltration rate (Kinner and Moody, 2010; Woods and Balfour, 2010). However, this relationship seems to only apply for a capillary barrier under dry conditions, as ash appears to conduct water better than the underlying soil under moist conditions, which may promote saturation overland flow or subsurface storm flow. On the contrary, if $K_{ash} < K_{soil}$, the ash layer limits the infiltration (Moody et al., 2009; Kinner and Moody, 2010);

(iii) wettability of ash can regulate the infiltration rate and, although this has not been quantified, Gabet and Sternberg (2008) detected its importance in runoff production.

2) The thickness of the ash layer. Its water storage capacity is proportional to its width (Cerdà and Doerr, 2008; Larsen et al., 2009; Woods and Balfour, 2010). Regardless of any pore clogging effect, the thicker the ash layer, the more likely it will be able to reduce or prevent runoff. However, once the ash layer is saturated the runoff rate depends on the intensity of the storm (Woods and Balfour, 2010).

Table 2. Delay of the onset of overland flow (min), overland flow coefficient (%) and total sediment yield (g cm^{-2}) from different rainfall simulation experiments comparing ash covered soil with bare soil.

Reference	Fire and forest information		Rainfall intensity (mm h^{-1}/ min of duration)	Plot size (m^2)	Soil texture and type		Mean depth (cm)	Delay of the onset of overland flow compared to bare soil (min)	Overland flow coeficient (%)		Total Sediment yield (g m^{-2})	
									Ash covered soil	Bare soil	Ash covered soil	Bare soil
Leighton-Boyce et al. (2007)	Wildfire, Eucalyptus plantations, Portugal		100/30	0.36	Loamy sand to sand, *Umbric Leptosols* and *Humic Cambisols*	Wettable soil	3.9	10 ± 3	0	13.2 ± 5.7	0	10.5 ± 5.8
						Water repellent soil (WDPT > 5h)		18 seconds	69.9 ± 9.3	99.7 ± 7.1	31.4 ± 5.8	114.7 ± 18.6
Woods and Balfour (2008)	High severity Wildfire, western Montana (USA)	1 month after the fire	75/60	0.5	Loam, *Typic Ustochrepts* from metasedimentary rocks		2.25	12 ± 10	16 ± 10	44 ± 12	274 ± 196	1040 ± 450
		9 months after the fire						-1.3 ± 1.8	28 ± 20	21 ± 8	414 ± 494	116 ± 80
		12 months after the fire						-2.2 ± 3.1	37 ± 14	37 ± 17	66 ± 66	< 66
Cerda and Doerr (2008)	High severity wildfire, *Pinus halepensis* forest Spain		55/60	0.25	Sandy loam, *Leptosols* and *Lithic Leptosols*		3.6	43	2.38	43.28	5	303.2
Zavala et al. (2009)	Low severity prescribed fire, open heathland, Spain		70/60	0.13	Loam		4.1	37 ± 13	4.2 ± 2	48 ± 18.8	2.6 ± 0.9	162.8 ± 200.3
Larsen et al. (2009)	High severity wildfire, *Pinus ponderosa* forest, USA. (Plots installed in the laboratory)		40/45	0.15	Granitic soil		0.5	3 ± 2	17 ± 3.1	35 ± 5.8	106	160
							1.2	6 ± 5	7.2 ± 0.8		73	
					Micaceous soil		0.5	2 ± 7	3.7 ± 0.7	7.8 ± 1.3	80	160
							1.2	6 ± 5	3.2 ± 1.4		66	

Reference	Fire and forest information		Rainfall intensity (mm h^{-1}/ min of duration)	Plot size (m^2)	Soil texture and type	Mean depth (cm)	Delay of the onset of overland flow compared to bare soil (min)	Overland flow coeficient (%)		Total Sediment yield (g m^{-2})	
								Ash covered soil	Bare soil	Ash covered soil	Bare soil
Woods and Balfour (2010)*	High severity controlled burns, *Pinus ponderosa* and *Pseudotsuga menziezii*	Immediately after the controlled fire	80/60	0.5	Sandy loam soil	1 cm white and black ash	3.8 ± 4.2	58 ± 28	16 ± 15	-	-
					Silt loam	1 cm black ash	10.3 ± 9.4	49 ± 30	58 ± 19		
		10 months after the controlled fire			Sandy loam soil	<0.1	3.5 ± 3.6	34 ± 16	16 ± 15		
					silt loam	<0.1	4.9 ± 6.3	47 ± 23	58 ± 19		
	Slash pile burn, *Pinus ponderosa* and *Pseudotsuga menziezii.* Ash addition to the plots				Sandy loam	0.5	-0.3 ± 5.9	49 ± 22	31. ± 22		
						2.5	7 ± 6.3	28 ± 7	45 ± 16		
						5	12 ± 8.4	17 ± 5	30 ± 25		

* Instead of onset overland flow it is considered time to ponding and instead of bare soil is pre-fire conditions.

3) The changes on the mineralogical properties of ash combusted at high temperatures containing CaO and $CaCO_3$. CaO can swell when wetted. If it swells prior to being introduced into the soil, its bigger size prevents it from blocking the pores in fine soils, but not in coarse ones (Gabet and Sternberg, 2008). If ash swells once incorporated into the soil, it further reduces the soil hydraulic conductivity (Etiegni and Campbell, 1991; Goforth et al., 2005). Ash containing $CaCO_3$ can create a material that can crust and compact, resulting in a very low hydraulic conductivity (this is a case of $K_{ash} < K_{soil}$) and Hortonian overland flow mechanism might be promoted (Figure 12; Balfour and Woods, 2006; Etiegni and Campbell, 1991; Onda et al., 2008; Woods and Balfour, 2008).

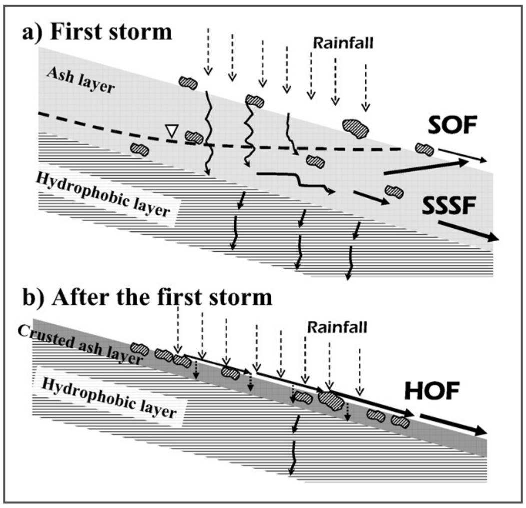

Figure 12. Schematic diagram of the runoff generation mechanisms of soil following fire (HOF: hortonian overland flow; SOF: saturaton overland flow; SSSF: subsurface storm flow). After Onda et al. (2008).

By controlling the soil hydrological behaviour, ash is thus able to also control erosion processes and sediment yield, increasing or reducing them according to the overland flow produced (Table 2). Despite the fact that Giovannini (1994)'s and Holcomb and Durgin (1979)'s experiments suggested that ash might be able to reduce soil erodibility by flocculating soil clay particles, no studies to date have confirmed this effect in the field. In contrast, it has been observed that under intense storms and irrespective of the nature of ash, it promotes related post-fire mass movements such as progressive debris flows (Bookter, 2006; Cannon et al., 2001). The mechanisms suggested by Burns (2007) and Gabet and Sternberg (2008) explain that when ash has delayed the onset of overland flow and is completely saturated, runoff can begin to flow downslope, incorporating additional ash and other fine particles. The fine particles increase the viscosity of the initial debris flow leading to a reduction in its settling velocity within the slurry, and causing the flow to become more dense, exerting a greater shear stress on the hillslope. In turn, the addition of the fine material decreases the settling velocity of the coarser material, thus further increasing the density of the flow and its erosivity. This mechanism, which evolves from an ash slurry into a debris flow, can deliver large volumes of sediment to valley floors.

1.3.3.4. Ash effects in water quality

Ash and nutrients lost off-site may affect downstream water quality. In many cases after an erosion event, the ash layer is only redistributed to the footslopes and in depressions (Cerdà, 1998a; Novara et al., 2011) but in others ash and solutes will reach streams and rivers. Reneau (2007) reported that in a reservoir in New Mexico with a mixed conifer catchment of 16.6 km^2, the ash yield the first year after fire was 2.1 Mg ha^{-1} (total 3567 Mg), which was 19% of the fine sediment exported and accounted for more than 90% of total ash exports. Even were only an small increase in post-fire sediment yield occurred during the first 15 months after a fire in the British Columbia, Petticrew et al. (2006) detected increases in the spatial and seasonal composition of the less than 500 µm composite suspended sediments, with more organic matter and black carbon.

The specific effect of ash on stream or reservoirs water quality is difficult to quantify given that the contribution of ash is rarely distinguished from that of mineral sediment delivered to streams (Smith et al., 2011). However, an increase in nutrients on streams and lakes due to ash after fires has been documented. For instance, Spencer et al. (2003) reported an increase from 5-60 fold in nitrogen and phosphorous above background levels in Montana. These nutrients arrived to the water bodies by volatilization followed by diffusion and dissolution of smoke into a stream or lake and atmospheric deposition of ash particulates and leaching in the water body (Lewis, 1974; Clayton, 1976; Raison et al., 1985; Spencer and Hauer, 1991; Spencer et al., 2003). Cations and nitrates usually arrive to the water by overland flow or by leaching of ash and subsequent movement with subsurface flow or through the entire soil profile to ground water (DeBano and Conrad, 1976; Soto and Diaz-Fierros, 1993; Williams and Melack, 1997; Hauer and Spencer, 1998; Gimeno-García et al., 2000; Lasanta and Cerdà, 2005).

The only study the author is aware of that focused specifically on the effects of ash on water quality is that of Earl and Blinn, 2003. This study in New Mexico involved an experimental ash input (1140 L ash slurry delivered over a 1.25 h period) to a first order stream, and monitoring of streams on burned catchments. The experimental ash input resulted in an immediate increase in concentrations of major cations and ammonium, nitrate and soluble reactive phosphate, as well as turbidity, conductivity and pH, while dissolved oxygen decreased. Changes in water chemistry were short-lived and returned to levels similar to before the experiment within 24 h except for the soluble reactive phosphate, which lasted for one month. In the burned catchments streams and in the catchments adjacent the burned ones, concentrations returned to pre-fire levels within 4 months. Other studies also found that nutrients returned to background concentrations within several weeks after the fire (Spencer et al., 2003; Lasanta and Cerdà, 2005). However, in some sites, nutrient concentration continued increasing periodically in subsequent years, especially during spring runoff and snowmelt (Tiedemann et al., 1978; Williams and Melack, 1997; Spencer et al., 2003).

Post-fire impacts on water quality may lead in some cases problems with maintaining the supply of potable water, although in many cases this only occurs for one storm event and turbidity peaks rapidly return to baseline conditions (Smith et al., 2011). However, this is one of the major issues of fire affecting urban areas after a wildfire. As an example, in the recent wildfires of Huesca and Catalunya in the North of Spain during March 2012, six days after fire the Regional Government informed the population near the catchments that the water quality was good, although in some areas there were turbulence problems and bottled water was needed for a few days (Puértolas, 2012). Notable in the context are for instance the disruption to the supply of drinking water of major cities such as Canberra and others cities in Victoria (Australia) from reservoirs within the burned catchment following the 2003 wildfire (White et al., 2006; Smith et al., 2011).

1.3.3.5. Ash in the carbon cycle

The combustion process in wildfires emits the carbon (C) stored in the biomass, litter and soil to the atmosphere mostly in form of CO_2, but also as carbon monoxide (CO) or methane (CH_4; van der Werf, 2006). The CO_2 and CO emissions to the atmosphere are balanced by uptake from surviving vegetation via regeneration. Therefore only fires that are associated with land use change and deforestation, i.e. those that are not followed by full biomass regeneration, are considered as net C emission events (Le Quére et al., 2009; van der Werf et al., 2009).

However, there are other ways of C storage after a wildfire as ash yields residues containing black carbon (Bowman et al., 2009). Black carbon (BC) is a by-product of burning with a range of chemistries that have a high resistance to biological and chemical degradation and hence long residence time in soils and other terrestrial sediments, and, ultimately, marine sediments (Forbes et al., 2006). Ash is a component containing parts of the black carbon continuum together with soot, charcoal, smoke and charred particulates (Forbes at al., 2006; Scott, 2010). In addition, although ash is not entirely comprised of organic components, other mineral constituents can contribute to C storage as well as calcium carbonate ($CaCO_3$).

To date, the role of ash has rarely been considered in studies examining C fluxes from wildfires, although it is known that black ash from different environments has a significant amount of organic content (Table 1). For instance, based on an ash bulk density of 0.4 g cm^{-3} (average of the reported in section 1.3.2) and a carbon content of 25% (Table 1), the organic carbon yield over the ground will range from 100 g m^{-2} to 2000 g m^{-2} (20 t ha^{-1}) for 1 and 20 mm ash depth respectively. Similarly Santín et al. (2012) using the same bulk density value, estimated a carbon deposition of 8.1 and 4.8 Mg ha^{-1} for sites affected by very severe and extreme burn severities in 2009 eucalypt forest fires near Melbourne. This demonstrates that that wildfires may actually lead to significant C sequestration, once the forest biomass has fully restocked. The authors visited the study site five month after the fire and much of the ash was found to be relocated to footslopes and depressions, as also observed Cerdà (1998a) and Novara et al. (2011). Ash mobilisation depends on its properties, such as particle size or density (Rumpel et al., 2009) and some of the ash will enter into the fluvial system and expected to form part of river, lake and, ultimately, marine deposits in the form of charcoal, one of the more resistant forms of C (Lal, 2003; Knicker, 2007). In any case, it would be protected to some degree from further erosion and consumption in future fires that might occur once a sufficient fuel load has been re-established. According to Santín et al (2012), this represents a true sequestration of C within the soil profile and in terrestrial or aquatic sediments. Hence there is a strong potential for BC to act as a significant sink of carbon from the more rapid bio-atmosphere carbon cycle to the slower (long-term) geological carbon cycle (Kuhlbusch and Crutzen, 1995; Forbes et al., 2006).

2. Rationale and objectives

2. Rationale and objectives

Fire is one of the most common ecological disturbances of the Mediterranean forest ecosystems (Arianoutsou et al., 1993; Lloret et al. 2009). During the long time of human occupation in the Mediterranean, humans developed their societies in coexistence with fire and using it to manage the land. Because the anthropogenic interventions have been present for a long time, most of the landscapes had reached some level of equilibrium between wildfire and human interaction through limiting vegetation growth by cutting the forest, controlled fire or grazing and using measures to control fire-derived problems such as soil erosion by terracing. However, cases of severe wildfires and associated degradation may have occurred (McNeill, 1992; Pyne, 2009).

Nowadays, with the extensive abandonment of Mediterranean agriculture, land is being transformed into unmanaged forest and scrublands (Lasanta and Vicente-Serrano, 2007) and fire is the key ecological factor that exerts the control of forest growth (Pausas et al., 2011). Since total suppression is not possible, citizens need to be aware of this fact and must learn to live with fire. After some intense fires, erosion can affect roads, water quality in reservoirs and rivers, and endanger farms, houses and lives. Therefore, the best way to live with fire is to know how it works, what its effects are and how to minimise its impacts.

Regarding the effects of wildfires, many studies have examined their impacts on soil physical, chemical and biological properties, soil hydrology and erosion, and vegetation although some research gaps remain for each topic. For instance, the variability of water repellency in calcareous Mediterranean forests and its prediction is a topic tackled very few times although water repellency may affect soil and hillslopes hydrology in fire-affected and long-unburned forests (Doerr et al. 2009). Also, very little is known about the effects of ash on soil hydrology (Martin, 2011) although the highest rates of overland flow and erosion, which can lead to water quality problems, usually arise in the first storms after a fire (Shakesby and Doerr, 2006; Moody and Martin, 2009), while ash is covering the soil and becomes a considerable part of the eroded sediment (Reneau, 2007). It is therefore of major importance to improve the knowledge and understanding of ash properties, and its dynamics and interactions with the soil and water. Ash hydrological properties have been analysed in only few studies given that its properties vary with plant types and fire severity. Also, the hydrological response and mechanism of runoff generation in a two layer system of ash and soil is poorly understood, adding to the fact that there are different ash types and soil might be wettable or water repellent. Furthermore, there are very few studies on the quality of overland flow from ash and how ash interacts with fine soil particles.

In order to contribute to a better understanding on the processes occurring immediately after a wildfire related to ash and its interaction with the soil system, with particular attention to wettability of ash and its effects on soil water repellency, the objectives of this thesis are to:

1) study the variability of water repellency at micro-, meso- and macro-scale in Mediterranean calcareous soils considering the vegetation type, soil depth and time since fire, and to attempt to predict its probability of occurrence;

2) report the ash dynamics after a wildfire and its duration on the ground identifying mechanisms of leaching through or/and incorporation into the soil, erosion by water or wind.

3) determine key properties of different types of ash including carbon content, particle size, hydraulic conductivity, water storage and specifically wettability;

4) examine the effects of ash incorporation into the soil on soil water repellency;

5) understand the effects of an ash layer covering the soil on the hydrological response of the system soil-ash and the mechanism of runoff generation considering ash layer thickness, ash type, soil texture, soil wettability and number of storms after a fire;

6) quantify erosion and sediment yield when an ash layer covers soil for various conditions of ash layer thickness, ash type, soil texture, soil wettability, and number of storms after a fire;

7) analyse the effects of an ash layer covering the soil on runoff water quality when an ash layer covers soil for various conditions of ash layer thickness, ash type, soil texture, soil wettability, and number of storms after a fire.

3. Resumen y discusión de los trabajos presentados

3. Resumen y discusión de los trabajos presentados

La repelencia al agua del suelo se ha documentado en un amplio rango de climas y bajo diferentes tipos de vegetación (Doerr et al., 2000). En el caso de los suelos calcáreos con pH mayor de 7 se entendía que no podían desarrollar repelencia al agua y, en efecto algunos suelos calcáreos mediterráneos como los *terra rossa* se ha comprobado que es muy complicado que la desarrollen, incluso cuando son afectados por el fuego (Mataix-Solera et al., 2008). Sin embargo, tanto esta Tesis Doctoral como trabajos recientes, han verificado que existe repelencia al agua en suelos mediterráneos calcáreos tanto quemados como no quemados (Mataix-Solera y Doerr, 2004; Arcenegui et al. 2007; Arcenegui et al., 2008; Tessler et al. 2008). La repelencia al agua es muy variable en el espacio, a micro, meso y macro escala según el tipo de vegetación y usos del suelo (Doerr et al., 2006; Mataix-Solera et al., 2007). Además, la repelencia al agua de los suelos depende de la estación y del tiempo transcurrido desde la última lluvia ya que la humedad es un factor fundamental que afecta a la repelencia. Otro factor que influye en la evolución de la hidrofobicidad es el tiempo desde el último incendio (Leighton-Boyce et al., 2005; Keizer et al. 2008; Malkinson y Wittenberg, 2011). Esta variabilidad espacial y temporal no está todavía cuantificada de forma exhaustiva en suelos calcáreos mediterráneos.

La presencia de la repelencia al agua en el suelo implica una reducción temporal de la capacidad de infiltración, que determina un incremento de la escorrentía superficial y flujos preferenciales subsuperficiales, y en zonas semiáridas es un factor fundamental en el balance hídrico (Doerr et al., 2000). A pesar de esta importancia, su predicción ha sido poco estudiada y los resultados obtenidos no son del todo concluyentes debido a su elevada variabilidad y cantidad de factores de los que depende, que todavía están en proceso de estudio (Scott, 2000; Doerr et al., 2006; Hubbert et al., 2006).

En el artículo original **"Spatial and temporal variations of water repellency and probability of its occurrence in calcareous Mediterranean rangeland soils affected by fires"** aceptado para su publicación en la revista Catena (ver anexo 6.2.; Bodí et al. 2012b), se procedió al estudio de las variaciones de la repelencia al agua a diferentes escalas espacio temporales en 4 cuencas afectadas por incendios forestales en diferentes momentos. Con los datos recogidos se realizaron además modelos estadísticos exploratorios para predecir la probabilidad de que estos suelos presenten repelencia al agua en función de una serie de factores estudiados. Esta última parte de modelización se realizó con la participación de Isabel Muñoz-Santa y Carmen Armero del Departament d'Estadística de la Universitat de València.

Las medidas de repelencia al agua y humedad del suelo se realizaron en 4 zonas de estudio en la Sierra de Enguera denominadas $F_1$99, $F_2$79, $F_3$08, $F_4$79, según el año en que ocurrió el último incendio, aunque todas ellas se quemaron en 1979. Cada zona comprendió aproximadamente 1 ha y se situaban a menos de 5 km de distancia unas de las otras. La repelencia al agua se midió en el campo utilizando el método denominado WDPT (del ingles, Water Drop Penetration Time) que consiste en registrar el tiempo que tarda en penetrar una gota de agua en el suelo y en función de los tiempos registrados las muestras se clasifican en 5 diferentes clases de persistencia de la repelencia al agua (Bisdom et al., 1993). La humedad del suelo se cuantificó gravimétricamente en el laboratorio. De esta manera, 10 gotas de agua se depositaron en parcelas de 10 cm × 10 cm debajo de 5 diferentes individuos de *Quercus coccifera, Pinus halepensis* y *Rosmarinus officinalis* y 5 parcelas de suelo desnudo (estas últimas sólo en $F_1$99 y $F_3$08), tanto sobre la superficie de suelo mineral como a un centímetro de profundidad. Los muestreos fueron mensuales y comenzaron inmediatamente desde el incendio de 2008, realizándose 18 campañas de medida y tomas de muestras entre Abril de 2008 y Julio de 2009, más un muestreo final en Agosto de 2011.

Los resultados indicaron que en total, más del 60% de las muestras de suelo eran hidrofílicas, aunque se encontró repelencia al agua en todas las zonas de estudio excepto en $F_1$99, incluso con valores de persistencia extrema. La zona de estudio $F_1$99 tuvo los valores más bajos de presencia de repelencia al agua (solo un 10%) tanto en superficie como en profundidad. En cambio, las dos áreas quemadas en 1979 ($F_2$79 y $F_4$79) tuvieron una presencia de la repelencia del 40% en superficie y 25% en profundidad. La zona recientemente quemada mostró valores de 17% de presencia en superficie y 27% en profundidad. No obstante, la presencia de la repelencia al agua varió a lo largo del año según la humedad del suelo. Los valores máximos de presencia de repelencia al agua del suelo de hasta un 60% y persistencia extrema ocurrieron en los veranos de 2008 y 2009 para $F_2$79 y $F_4$79. Sin embargo, la zona $F_3$08 recientemente quemada, tuvo un elevada repelencia el verano del incendio (2008) pero disminuyó al siguiente hasta un 10%, tanto en superficie como en profundidad.

En general, estos valores son similares a los medidos en zonas mediterráneas calcáreas y más bajos que en brezales en el sur de España, bosques de eucaliptos o bosques mixtos del norte de Europa y América (Buczko et al., 2005; Doerr et al., 2009; Martínez-Zavala y Jordán-López, 2009; Stoof et al., 2011). Sin embargo, en este trabajo, la repelencia al agua fue distinta según las zonas de estudio. Inmediatamente tras el incendio los suelos tenían valores de repelencia al agua similares a los de las zonas quemadas en 1979, aunque más elevados en profundidad seguramente inducidos por el fuego al condensarse las sustancias hidrofóbicas en capas inferiores del suelo (DeBano et al., 1970). No obstante, un año tras el incendio la repelencia al agua del suelo desapareció, lo que confirma los hallazgos de otros investigadores en el mediterráneo (Tessler et al., 2008). Las observaciones de campo indicaron que la desaparición de la repelencia se debió a la falta de aportes de nueva hojarasca con sustancias frescas inductoras de hidrofobicidad combinado con la deposición de suelo y sedimentos erosionados de la parte alta de la laderas que cubrieron el suelo original, en algunos casos hasta 5 cm (Doerr et al., 2000; Hubbert y Oriol, 2005). El sedimento estaba compuesto de cenizas, de las cuales el 88% eran hidrofílicas (Bodí et al., 2011b; ver anexo 6.3.). Tres años más tarde, en 2011, esta zona de estudio continuó con una repelencia baja y no está claro cuando la repelencia volverá a tener los valores iniciales como los de las zonas quemadas 20 años atrás ($F_2$79 y $F_4$79). Asimismo, la zona quemada 10 años antes ($F_1$99) todavía tenía valores bajos de repelencia al agua. Este patrón de dinámica de la repelencia al agua en el suelo a largo plazo tras un incendio coincide con el modelo conceptual que elaboraron Malkinson y Wittenberg (2011).

Respecto a las variaciones a meso escala, no se encontraron diferencias en la presencia y persistencia de repelencia al agua del suelo debajo de *Quercus coccifera, Pinus halepensis* y *Rosmarinus officinalis* en ninguna de las zonas de estudio. Solamente el suelo desnudo fue significativamente menos repelente que debajo de las especies vegetales estudiadas tanto en superficie como en profundidad. A pesar de que en otros trabajos en zonas calcáreas mediterráneas se han encontrado diferencias entre especies, siendo debajo del *Pinus halepensis* donde se exhibe más repelencia (Arcenegui et al., 2007; Mataix-Solera et al., 2007; Tessler et al., 2008), en este estudio no se observaron debido a la gran variabilidad de la repelencia debajo de los individuos de la misma especie. La persistencia de la repelencia varió hasta en 4 clases tanto en la misma especie vegetal como entre especies. La variabilidad fue menor en $F_1$99 ya que la mayoría de muestras eran hidrofílicas y, también descendió en cuanto la humedad aumentó, pasando a ser hidrofílicas el 100% de las muestras cuando superaron el umbral de humedad de 23,3% en la zona $F_1$99, 42,7% en $F_2$79, 56,3% en $F_3$08 y 45,1% en $F_4$79. La amplia variabilidad que enmascara la diferencia entre especies puede deberse a la homogénea cobertura del suelo (en mayor o menor grado según la zona de estudio) formada por de hojarasca de distintas especies superpuestas, junto con un discontinuo y todavía por sistematizar patrón de distribución de la humedad del suelo, microorganismos e impacto del calentamiento del suelo tras el incendio (Rodriguez-Iturbe et al., 1995; Cerdà, 1997; Bochet et al., 1999; Spielvogel et al., 2009; Gimeno-García et al., 2011).

De igual modo, a escala milimétrica, la disposición de microorganismos y de composición de la materia orgánica no es continua (Hallett et al., 2004) pudiendo incluso cambiar dentro de los agregados del suelo (Urbanek et al., 2007). La distribución de estos elementos aparentemente aleatoria refleja también los resultados de repelencia al agua del suelo obtenidos a micro escala con parcelas de 10 cm x 10 cm entre las 10 WDPT mediciones. En este caso la variabilidad fue igual de grande que entre especies y entre individuos de la misma especie. Ninguna especie mostró menos variabilidad que otra excepto el suelo desnudo donde la mayoría de las muestras eran hidrofílicas. Esta variabilidad fue de nuevo menor en $F_1$99 y especialmente cuando los valores de humedad eran altos.

Con estos datos, utilizando como covariable el contenido de humedad y como factores de efectos fijos el tipo de vegetación bajo el que se realiza la medida (*Quercus coccifera, Pinus halepensis* y *Rosmarinus officinalis*) y la profundidad (superficie, 1 cm), se realizó un modelo para predecir la probabilidad de la presencia de la repelencia al agua del suelo (es decir, si está presente o no) en ambas zonas de estudio quemadas en 1979 (F_279 y F_479) en las cuales la repelencia al agua es regular a largo plazo. El modelo se llevó a cabo con el programa R y es del tipo “Logístico de Regresión de Efectos Mixtos”, dentro de los “Modelos Lineares Generalizados” (Hedeker, 2005; Faraday, 2006). Este modelo permitió trabajar con distribuciones no normales y, además de utilizar covariables y factores fijos, considerar factores aleatorios, incluyendo las características del muestreo aleatorio realizado, y las relaciones entre factores, por ejemplo entre superficie y profundidad, asegurando que si existen diferencias entre estas son debidas a la profundidad y no a la diferencia entre las parcelas. Se empezó con el modelo más complejo introduciendo todas las covariables, factores fijos y relaciones posibles y se fueron obteniendo modelos más simples comparando, seleccionando y quitando o transformando para el siguiente paso los menos significativos. Este proceso se continuó hasta que se obtuvo el mejor y más simple modelo con el mínimo AIC y BIC (del inglés Akaike Information Criteria y Bayesian Information Criterion). Ambos parámetros describen la bondad del modelo y su complejidad entre modelos obtenidos a partir de los mismos datos. Tras el primer análisis se comprobó que aunque las dos zonas quemadas en 1979 son parecidas, no lo son sus interacciones entre factores y por tanto se trataron separadamente para obtener mejores resultados.

El modelo resultante para la zona F_279 comprendió como mejores variables explicativas ($P < 0{,}0001$) el factor fijo profundidad (superficie o 1 cm) y la covariable humedad del suelo. No se obtuvo una diferenciación de la probabilidad de presencia de la repelencia entre los diferentes tipos de vegetación, al igual que también ocurría anteriormente con la persistencia. Así, la probabilidad calculada de obtener repelencia al agua en un suelo resultó ser mayor en superficie y conforme disminuye la humedad. La desviación estándar asociada a todas las parcelas en conjunto y la asociada a cada parcela en superficie y profundidad fue elevada, indicando de nuevo la variabilidad entre las parcelas bajo cada individuo de cualquier especie. Para la zona estudio F_479 el modelo resultó algo más complejo incluyendo como mejores factores explicativos ($P < 0{,}0001$) la humedad del suelo y el tipo de vegetación, aunque sólo diferenciando entre el *Pinus halepensis* y el resto. Otros valores explicativos significativos ($P < 0{,}001$) también fueron la profundidad y la interacción entre la humedad y la especie. La probabilidad de repelencia calculada en el modelo indicó que en esta zona de estudio es mayor bajo *Pinus halepensis*, en superficie y tanto más cuanto menor sea la humedad del suelo. En este caso las desviaciones estándar fueron la mitad que en el modelo de la zona F_279, mostrando una menor variabilidad entre las parcelas.

El modelo obtenido es potente en el contexto aplicado, ya que tiene una alta capacidad de estimar probabilidades a partir de las covariables utilizadas, y podría obtenerse en otros lugares -siguiendo el mismo procedimiento- donde la vegetación haya alcanzado un estado maduro estable. Se sugiere que los enfoques bayesianos mejorarían los resultados así como la incorporación de otras variables fáciles de obtener como el grosor de la capa de hojarasca. Además, para estimar la probabilidad de la presencia de la repelencia o incluso la persistencia en sitios variables en el tiempo como los afectados por incendios forestales, habría que utilizar un modelo dinámico donde se tuvieran en cuenta factores tales como el movimiento e incorporación de cenizas y sedimentos al suelo y la recuperación de la vegetación, microorganismos e incorporación de material orgánico hidrofóbico, junto con sus relaciones con la repelencia al agua del suelo. No obstante, estos datos todavía necesitan un mejor estudio, sobre todo si el objetivo es incorporarlos a un modelo estadístico de predicción.

Con el propósito de aumentar el conocimiento sobre la dinámica de las cenizas y su efecto en la repelencia al agua del suelo y en los procesos hidrológicos en general tras un incendio, se llevaron a cabo diferentes trabajos. El primero, publicado en la revista Geoderma "**The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content"** (ver anexo 6.3.; Bodí et al. 2011b) tuvo como objetivos examinar y cuantificar la repelencia al agua en las cenizas y sus efectos en el suelo. Aunque las cenizas se han considerado en general un material altamente hidrofílico por su capacidad de almacenar agua (por ejemplo Cerdà y Doerr, 2008; Etiegni y Campbell, 1991; Woods y Balfour, 2010), en otras descripciones de estudios científicos, este hecho no se confirma tan claramente (Gabet y Sternberg, 2008; Khanna et al., 1994; Stark, 1977). Es por tanto que en este estudio se cuantificó por primera vez la repelencia al agua de cenizas y se comprobó su relación con el contenido de carbono orgánico total (TOC en inglés) y el color, ya que ambos son indicadores del grado de combustión de la materia vegetal. Las medidas se realizaron en muestras recogidas en 5 incendios forestales en la cuenca mediterránea quemados con distinta severidad (en Israel y en la Península Ibérica en las provincias de Valencia y Alicante) y con muestras producidas en el laboratorio a diferentes temperaturas: 200, 250, 300, 350, 400, 450, 500, 550 y 700 °C durante 20 min a partir de vegetación similar a la presente en los incendios forestales muestreados, concretamente de hojarasca de las siguientes especies: *Quercus coccifera, Pinus halepensis* y *Rosmarinus officinalis.* Además, con el objetivo de comprobar los efectos de las cenizas en la repelencia al agua tras incorporarse en el suelo, las cenizas producidas a 250, 350, 500 y 700 ºC se mezclaron en diferentes dosis (4% y 16%, equivalentes a 0,5 cm y 2 cm sobre el suelo respectivamente) con el mismo suelo siendo hidrofílico y repelente al agua (WDPT > 3600 s) tras calentarlo en el horno mufla. También se comprobó la repelencia al agua tras humedecer la muestra con 20 mL de agua y dejarla secarse al aire.

Los resultados mostraron inequívocamente que en efecto existen cenizas repelentes al agua, concretamente el 33% de las muestras recogidas en incendios forestales. Sin embargo hubieron incendios con mayor o menor proporción de cenizas repelentes, desde 13% en un incendió donde llovió previamente a la recolección de las cenizas hasta un 60% de las muestras en el incendio de menor severidad. Los resultados de laboratorio complementaron los obtenidos en el campo e indicaron también que las cenizas generadas a bajas temperaturas (200 y 250 ºC), en el caso de *Quercus coccifera* y *Pinus halepensis,* exhiben repelencia extrema y que disminuye entre 250 y 400 ºC. Las muestras de *Rosmaninus officinalis* no mostraron repelencia a 200 ºC pero se incrementó a 250-350 ºC sin llegar al grado de repelencia severa como en las otras especies. A temperaturas superiores a 400 ºC la repelencia desapareció en las tres especies. Durante los años de realización del doctorado se han encontrado más muestras de cenizas de incendios forestales con repelencia al agua en Colorado (resultados no publicados), Portugal (González-Pelayo, 2009) y en incendios prescritos en Victoria (Australia; resultados no publicados y comunicación personal de Petter Nyman) indicando que cenizas de *Pinos sp.* y *Eucaliptos sp.* son potencialmente repelentes al agua.

Debido a que en la naturaleza los compuestos hidrofóbicos son orgánicos (Doerr et al., 2000), es lógico el análisis de la relación de la repelencia al agua de las cenizas y su contenido en carbono orgánico total (TOC). En las muestras de incendios forestales, el TOC osciló entre 4,6% y 31,1%. Los incendios de menor severidad mostraron los mayores valores de TOC ya que conforme aumenta la temperatura de combustión los compuestos orgánicos desaparecen. Estos compuestos orgánicos procedentes de materiales como la madera, la hojarasca o la materia orgánica del suelo, asociados con combustiones de baja severidad, entre 250-500 ºC, pueden producir substancias hidrofóbicas al eliminar los grupos funcionales oxigenados e incrementar los aromáticos (Baldock y Smernik, 2002; González-Vila et al., 2009; Atanassova y Doerr, 2011). Es por tanto que la repelencia al agua también fue mayor en los incendios de menor severidad con una relación moderada entre el TOC y la repelencia al agua ($r = 0,7966$, $p < 0,01$). La existencia de los compuestos orgánicos hidrofóbicos se demostró en estas muestras gracias a la colaboración con profesor Pavel Dlapa de la Comenius University de Bratislava, República Eslovaca, y que ha derivado en otra publicación científica (ver Dlapa et al., 2012). Con el análisis de Espectroscopía Infrarroja Transformada de Fourier, se comprobó que la repelencia en las cenizas también está íntimamente relacionada con compuestos alifáticos, aromáticos y carboxílicos, pero además, con la relación entre el carbono orgánico e inorgánico que en las cenizas repelentes que fue más del doble. Respecto al color, las cenizas fueron también más claras conforme disminuye el contenido de carbono orgánico por la combustión, pero sin embargo la relación entre el valor Munsell y el TOC y el valor Munsell y la repelencia al agua no fue muy buena ($r = 0,5702$, $p < 0,01$ y $r = 0,5930$, $p < 0,01$, respectivamente). La causa puede ser que los tonos de colores cambian según el incendio forestal, hecho que sugiere que el color cuantificado con la tabla Munsell puede no ser un indicador de la severidad demasiado exacto para comparar entre incendios, aunque utilizado a grandes rasgos (en términos de cenizas blancas o negras) si que informa de la severidad el incendio.

Las muestras de laboratorio mostraron unos valores de TOC mayores que las de campo, de 22,9% a 60,3%, muy elevados para temperaturas de hasta 700 ºC. La relación con la repelencia fue baja ($r = 0,3493$, $p < 0,01$), igualmente que las relaciones entre el TOC, la repelencia al agua y el color fueron débiles ($r = 0,2257$, $p < 0,01$ para el valor Munsell y TOC y $r = 0,1992$, $p < 0,01$ para el valor Munsell y la repelencia al agua). En estas muestras además no se observó ninguna tendencia gradual de colores de oscuro a claro a menores y mayores temperaturas de combustión respectivamente. Los colores de las muestras se alternaron conforme la temperatura aumenta sin ningún patrón claro aparente, sobre todo en las muestras de *Rosmarinus officinalis.* Tanto los mayores valores de TOC como la alternancia en los colores, se cree que fue debida a que los experimentos de laboratorio no reflejan en ocasiones demasiado bien las condiciones de combustión de un incendio forestal. Así, se observaron ciertos efectos que pudieron haber afectado a este proceso de combustión y modificado las temperaturas determinadas en el horno mufla: (i) una continuación de la combustión después de sacar la muestra de la mufla al oxigenarse, y (ii) una mayor combustión en el centro de las muestras respecto a la superficie y los márgenes. Estos efectos similares, así como el poco flujo de aire en el horno también han sido indicados por Raison (1979), Misra et al. (1993) y Gray y Dighton (2006). Se recomienda por tanto, comparar estudios de cenizas creadas en el laboratorio con estudios de cenizas recogidas en el campo.

Bodí et al. (2011b) también demostraron que la incorporación en el suelo de cenizas producidas en el laboratorio modifican su humectabilidad según la naturaleza de ambos materiales. El suelo hidrofílico al que se le incorporó cenizas repelentes de *Quercus coccifera* y *Pinus halepensis* quemadas a 250 ºC y *Rosmarinus officinalis* a 350 ºC aumentó su repelencia al agua a valores de repelencia fuerte en dosis altas y repelencia baja en dosis bajas. Las cenizas repelentes todavía aumentaron más la repelencia al agua en el suelo ya hidrofóbico. Por otra parte, las cenizas hidrofílicas (producidas a 500 y 700 ºC) redujeron la repelencia al agua de los suelos hidrofóbicos, aunque no la hicieron desaparecer, tal vez por la elevada repelencia inicial del suelo. Cuando las muestras se humedecieron y se secaron, como tras una lluvia, la repelencia al agua desapareció en la mayoría de muestras, excepto en el suelo repelente con cenizas repelentes y el suelo hidrofílico con cenizas de *Quercus coccifera* producidas a 250 ºC. Por tanto, el efecto de las cenizas cuando se incorporan al suelo puede tener repercusiones hidrológicas al aumentar o reducir la repelencia suelo. El aumento puede producirse sobre todo si las cenizas se incorporan por bioturbación o son arrastradas por el viento ya que cuando se humedecen se ha evidenciado que reducen su repelencia. Este hecho también se ha comprobado en el campo, ya que las cenizas que tienen en este estudio menor repelencia al agua debido a una lluvia posterior al incendio se recogieron en el mismo incendio del estudio anterior y contribuyeron a reducir la repelencia en la zona quemada al incorporarse al suelo (anexo 6.2.; Bodí et al. 2012b).

Tras comprobar que existen cenizas hidrofóbicas y que cuando las cenizas en general, tanto hidrofílicas como repelentes al agua, se incorporan al suelo pueden modificar la hidrofobicidad de éste, se procedió al estudio de sus efectos antes de su incorporación, es decir cuando todavía cubren el suelo. Se cuantificó como afecta una capa de cenizas a la repelencia al agua del suelo y a la producción de escorrentías y erosión y, así se pudo además observar como se produce su incorporación tras la precipitación. Con este objetivo se realizó la estancia de investigación de cuatro meses en la Swansea University (Swansea, Reino Unido) y se escribió la siguiente publicación, la cual ya está aceptada y en prensa en la revista Geoderma **“Hydrological effects of a layer of vegetation ash on underlying wettable and water repellent soil”** (ver anexo 6.4.; Bodí et al., 2012a). Conocer el comportamiento hidrológico de la interacción de la capa de cenizas que cubre el suelo (o en un sistema de dos capas como lo denominan Kinner y Moody, 2010) durante los eventos de precipitación es decisivo para saber como serán las escorrentías y la pérdida de suelo inmediatamente tras un incendio. Para ello, se llevó a cabo un experimento de lluvia simulada en el laboratorio con un simulador de tipo goteador utilizado por Leighton-Boyce et al. (2007) y Terry y Shakesby (1993). La intensidad de lluvia aplicada fue de 82,5 mm h^{-1} durante 40 min sobre unas parcelas cuadradas de 0,09 m^2 con una pendiente del 10º. Estas parcelas fueron diseñadas para recoger la escorrentía superficial y subsuperficial y cada lado fue provisto de un colector para almacenar y medir la erosión por salpicadura (Terry, 1989). Unas parcelas se prepararon con una capa de 3 cm de suelo hidrofílico y otras con el mismo suelo pero repelente al agua (WDPT > 3600s), la cual fue inducida mediante un proceso de calentamiento húmedo con el autoclave. La inducción de la repelencia al agua en el suelo mediante un proceso de autoclavado se investigó con la colaboración con Emilia Urbanek de la Universidad de Swansea y ha derivado en una publicación en la revista Soil Science Society of America Journal (ver Urbanek et al., 2010). El suelo se recubrió con una capa de cenizas, recogidas en un incendio forestal de baja severidad en Teruel, con diferentes espesores (5, 15 y 30 mm) y otras parcelas se dejaron sin recubrir como control. Para monitorizar como se producía la humectación del suelo se instalaron además dos sensores de humedad a 1,5 cm de profundidad del suelo. Las lluvias simuladas imitaron tres situaciones distintas de eventos de lluvia. Todas las muestras se sometieron a una primera lluvia pero una parte se expuso a una segunda lluvia a las 24 h y otra parte a una segunda lluvia tras 4 días secándose a 25 ºC. Durante cada experimento se midió la escorrentía cada minuto y cada 5 minutos se determinó la producción de sedimentos. Después de cada simulación se abrieron los perfiles del suelo para explorar el frente de humedad y tomar muestras para su análisis.

Los resultados indicaron que durante la primera simulación de lluvia, el suelo no repelente sin cubierta de cenizas absorbió el 100% de la lluvia pero un 57% salió del sistema como escorrentía subsuperficial ya que la cantidad de agua fue mayor que la capacidad de retención hídrica del suelo. En las parcelas cubiertas por cenizas se produjo una ligera cantidad de escorrentía superficial que aumentó con el grosor de la capa de cenizas y alcanzó como máximo un 4% de la lluvia cuando el espesor de cenizas fue de 30 mm de profundidad. La escorrentía se retrasó y se prolongó por más tiempo conforme aumentó el espesor de la capa de cenizas, pero en todos los casos desapareció durante la lluvia. Cuando los experimentos se realizaron sobre suelo repelente al agua se produjeron efectos opuestos. El coeficiente de escorrentía superficial fue el 78% para el suelo desnudo, 77% para la parcela de suelo cubierto con una capa de cenizas de 5 mm de espesor, 50% para la cubierta con 15 mm y 26% con 30 mm. Es decir, el coeficiente de escorrentía se redujo conforme el espesor de la capa de cenizas aumentó. La escorrentía en ambos casos, tanto en el suelo hidrofílico como en el hidrofóbico, empezó en el mismo momento para cada grosor de la capa de cenizas, aunque en el suelo repelente al agua fue continua hasta el final de la simulación de lluvia. Esto indica que la escorrentía apareció cuando la capa de cenizas se saturó de agua, y que el espesor de las cenizas determina la generación y el volumen de la escorrentía, al aumentar la capacidad de retención hídrica (Larsen et al., 2009; Woods y Balfour, 2010). La cantidad de agua calculada que pueden almacenar estas cenizas oscila entre el 60-80% de su masa y coincidió con la porosidad de las cenizas, un 80%. Estos datos concuerdan con otros calculados por Cerdà y Doerr (2008), Woods y Balfour (2008) y Zavala et al. (2009).

El hecho de que en el suelo hidrofílico se produjera escorrentía cuando las cenizas se saturaron y desapareciera durante la lluvia simulada con mayor duración conforme el grosor de la capa de cenizas fue mayor, sugiere que las cenizas absorben el agua desde la interfaz con el suelo, donde producen encharcamiento. Es decir, el proceso es de abajo a arriba, y no al contrario como se produce la infiltración en los suelos (Cerdà, 1996). Una vez saturadas las cenizas, la arroyada superficial fluye hasta que el suelo consigue drenar el agua que las cenizas ya no pueden absorber además del agua de la lluvia que todavía no había cesado, es decir, hasta que las dos capas se equilibran. Si consideramos individualmente la capa de cenizas, la escorrentía que se produce fue por exceso de saturación, pero si observamos el suelo, la escorrentía en este caso temporal, se produce por exceso de infiltración. Efectos similares han sido discutidos por Kinner y Moody (2010) donde se sugiere que la ceniza actúa como una barrera capilar hasta que se satura. En el suelo repelente, la escorrentía empezó en el mismo momento que en el suelo hidrofílico pero debido a su casi nula capacidad de infiltración continuó hasta el final de la lluvia. En este caso, la escorrentía superficial no solo se redujo porque la capa de cenizas retrasa y almacena agua si no porque esta capa también ejerce presión hidráulica en el suelo y fuerza el contacto del agua con el suelo reduciendo la repelencia en las zonas mojadas. En efecto, la forma resultante del hidrograma en las parcelas con 15 y 30 mm de capa de cenizas resultó ser la clásica planteada por DeBano (1981) donde la capacidad de infiltración aumenta conforme el tiempo de lluvia aumenta. El aumento de la capacidad de infiltración de estos suelos extremadamente repelentes en inicio se confirmó además con la aparición de escorrentía subsuperficial y la mayor humedad del suelo comparado con el suelo desnudo y 5 mm de cobertura de cenizas. Asimismo, en la observación del perfil se detectaron flujos preferenciales a lo largo de éste que actúan como vías por donde el agua registrada como flujo subsuperficial transcurre (Ritsema y Dekker, 1994).

Antes de la segunda lluvia se observó en la superficie de la parcela una redistribución de cenizas hacia la parte baja, aunque el suelo todavía estaba completamente cubierto de cenizas con algunas marcas del impacto de la lluvia. En las parcelas que se abrieron se observó una mezcla de los dos materiales sólo en la interfaz entre de las cenizas y el suelo. Tras la segunda lluvia, tanto 24 h después de la primera como 4 días después, el suelo hidrofílico sin recubrimiento de cenizas tampoco produjo escorrentía, pero el flujo subsuperficial aumentó especialmente 24 h después de la primera lluvia ya que el suelo todavía seguía completamente húmedo. En cambio, en las parcelas recubiertas por cenizas se produjo más escorrentía superficial que en la primera simulación, siendo 14% el coeficiente de escorrentía tras 24 h y 17% tras 4 días sin diferencias significativas entre espesores de la capa de cenizas. El motivo de esta producción de escorrentía se cree que fue debido a el bloqueo de los poros del suelo por la cenizas que se hincharon tras estar en contacto con el agua tras la primera lluvia (Etiegni y Campbell, 1991; Stoof et al., 2010). Además, este suelo es de textura gruesa (franco arenosa) y con una estructura artificial ya que fue puesto en la parcela tras ser tamizado, por tanto susceptible a este fenómeno (Woods y Balfour, 2010). En este caso el mecanismo de producción de escorrentías para el sistema de cenizas y suelo fue por el exceso de infiltración en el suelo (aunque para las cenizas solo sea por saturación), donde el suelo cubierto y taponado por las cenizas no es capaz de infiltrar la lluvia a la intensidad que se produce.

En el suelo repelente al agua no puede evidenciarse si ocurrió taponamiento de poros o no ya que la escorrentía generada por la repelencia al agua del suelo enmascaró si también se produjo escorrentía por otros motivos. En cualquier caso, igual que en la primera lluvia, la escorrentía empezó a la vez que en el suelo hidrofílico en ambas segundas lluvias, es decir, cuando las cenizas se saturaron. Sin embargo, al contrario que en el suelo no repelente, el coeficiente de escorrentía se redujo respecto la primera lluvia simulada para el suelo desnudo y las cubiertas con 15 y 30 mm de ceniza (a un 20% de coeficiente de escorrentía superficial) aunque los 5 mm de capa de cenizas escurren todavía un 70% de la lluvia. A pesar de la disminución general de la escorrentía, en este caso las cenizas no fueron tan relevantes en la reducción de la repelencia al agua del suelo y el incremento de su capacidad de infiltración como en la primera lluvia, porque el propio suelo ya húmedo en superficie hizo el mismo efecto que hacían las cenizas en el primer evento. Al contrario, los 5 mm de cenizas aumentaron el coeficiente de escorrentía superficial. Se cree que el delgado espesor de cenizas redujo el contacto del agua con el suelo y no ejerció suficiente presión sobre suelo para humedecerlo y por tanto siguió siendo más repelente que el mismo suelo desnudo. En este caso, sólo el suelo desnudo tuvo la forma del hidrograma típico de un suelo repelente con reducción final de la tasa de escorrentía. En la segunda simulación de lluvia en la que el suelo estuvo 4 días secándose, el coeficiente de escorrentía se redujo todavía más en suelo desnudo (17%) y también con 5 mm (a 30%) pero aumentó en 15 y 30 mm a 30%, con un hidrograma también típico de suelo repelente en todos los casos, mostrando de nuevo que las cenizas tras la primera lluvia no promueven la reducción de la repelencia al agua en mayor medida que el suelo desnudo. El aumento de la repelencia en el suelo desnudo (WDPT = 950 s) tras el periodo de secado junto con la forma del hidrograma, indica que la repelencia al agua se restableció también en el suelo cubierto de cenizas y fue reduciéndose durante la lluvia. El perfil de suelo resultante de ambas segundas lluvias mostró evidencias de humedecimiento y reducción de la repelencia del suelo mayores que tras la primera lluvia junto con mayor cantidad de escorrentía subsuperficial a través de flujos preferenciales, sobre todo en el caso de 15 y 30 mm de cobertura de cenizas. En el caso de los 5 mm de cenizas, solo resultó completamente humectable el primer centímetro debajo de las cenizas y el resto del suelo siguió teniendo repelencia extrema (WDPT > 3600s).

La cantidad de sedimentos producidos tras cada lluvia únicamente superó los 66 g m^{-2} en las parcelas con suelo repelente tanto en la primera y como en la segunda lluvia 24 horas después de la primera. En los casos de la primera lluvia en el suelo repelente, la cantidad de sedimentos producidos fue mayor en suelo desnudo (300 g m^{-2} aprox.), que resultó ser además la única parcela donde la erosión por salpicadura fue significativa;

seguida por 5 mm de cobertura, 15 y 30 mm (esta última con 100 g m^{-2} aprox.). Durante la segunda lluvia simulada se produjo una erosión considerable (aproximadamente 340 g m^{-2}) en todas las parcelas cubiertas por cenizas. El suelo desnudo esta vez no produjo apenas sedimento ya que estaba húmedo (mayor cohesión), sin repelencia al agua y por tanto pasó a ser menos erosionable. En el caso de 15 y 30 mm de espesor capa de cenizas es importante notar que el sedimento producido no fue proporcional a la escorrentía como en todos los otros casos ($r = 0{,}941$, $p < 0{,}05$) y por tanto parece ser que se las cenizas fueron capaces de aumentar la capacidad de transporte del sedimento produciendo un proceso de flujo extraordinario de sedimentos similar a de una colada de derrubios descrito por Burns (2007) y Gabet y Sternberg (2008).

Con el objetivo de conocer cual es el comportamiento en la escorrentía y sedimento producido en estas mismas cenizas recubriendo otro tipo de suelo y además compararlo con otro tipo de cenizas se llevó a cabo un nuevo experimento con lluvia simulada. En este estudio se analizó además la composición química de las escorrentías (pH, conductividad eléctrica, Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}, Cl^{-} y NO^{3-}) con la intención de encontrar las diferencias respecto a la propia composición de las cenizas, parte mucho más estudiada, y comprobar sus efectos en la calidad del agua. Este trabajo se realizó junto con el estudiante de doctorado de la Universidad de Zaragoza Francisco Javier León Miranda y en este momento se están escribiendo varios artículos científicos. Asimismo, el estudio se presentó en Febrero de 2011 en el III congreso Intenacional "Fire Effects on soil properties" en Guimaraes (Portugal) con el título: **“Runoff rates, water erosion and water quality from a soil covered with different types of ash”** (ver anexo 6.5.; Bodí et al., 2011a). En este caso la intensidad aplicada fue de 55 mm h^{-1} durante 1 hora y el simulador utilizado fue del tipo pulverizador (Cerdà et al., 1997). La simulación de lluvia se realizó en el campo, sobre suelos sin alterar y delimitados por parcelas circulares de 0,22 m^2 en una zona con una pendiente promedio de 5º. Las parcelas se cubrieron con las mismas cenizas que el experimento anterior (cenizas recogidas en Teruel en un incendio forestal se baja severidad) y con cenizas producidas en el laboratorio a alta temperaturas de *Citrus sp*. La estructura del experimento fue similar a la anterior en cuanto la lluvia simulada se realizó sobre suelo desnudo y sobre suelo cubierto con una capa de cenizas de 5, 15 y 30 mm de grosor. También se realizó una primera lluvia y segunda lluvia, pero a diferencia del trabajo anterior, todas mas muestras se sometieron a esta segunda lluvia una semana después, tiempo en el que las parcelas quedaron a la intemperie igual que en situaciones naturales. Las muestras de escorrentía se midieron cada minuto y se recogieron cada 5 minutos para determinar el sedimento y la composición química del agua.

Los resultados de este trabajo indicaron que este suelo desnudo tenia una menor capacidad de infiltración que el suelo hidrofílico utilizado en el experimento anterior en el laboratorio. Con una intensidad menor (55 mm h^{-1} comparado 82,5 mm h^{-1}) produjo un 2% de coeficiente de escorrentía medio, indicando así también una menor conductividad hidráulica asociado en algunos casos a pequeñas manchas de repelencia al agua ligera en algunas parcelas y compactación del propio suelo. Cuando éste se cubrió con cenizas recogidas en el incendio forestal en Teruel, las mismas que las utilizadas en el experimento anterior, produjeron una escorrentía de no más del 12% (en las parcelas cubiertas por 5 mm de cenizas), pero de manera contraria al experimento anterior, disminuyó conforme aumentó el grosor de la capa de cenizas ya que la escorrentía continua hasta el final de la lluvia. La continuidad de la arroyada se debe a la menor capacidad de infiltración del suelo, y como en el anterior experimento se cree que el sistema cenizas-suelo acabó equilibrándose y adoptando las tasas de escorrentía similares a las del suelo sólo (nulas en el experimento anterior), promovida primero por exceso de infiltración en el suelo y cuando se equilibra por exceso de saturación.

Por otra parte, el suelo cubierto por cenizas producidas a alta temperatura en el laboratorio produjeron mayor cantidad de escorrentía que el suelo desnudo o con el tipo de cenizas producidas a baja severidad. El coeficiente de escorrentía siguió siendo también menor conforme el espesor de la capa de cenizas aumentó, 40% en 5 mm de espesor, 23% en 15 mm y 12% en 30 mm y empezó de 2 a 4 minutos antes que las parcelas cubiertas con las cenizas recogidas en el incendio de baja severidad. Sin embargo, las cenizas producidas a alta temperatura presentaron una mayor porosidad y conductividad hidráulica (68% y K_{sat} = 304 mm h^{-1}) que las cenizas recogidas en el incendio (62% y K_{sat} = 138 mm h^{-1}). Por tanto, el motivo de esta mayor producción de escorrentía se cree que fue causada porque estas cenizas se compactaron por el efecto del impacto de la lluvia debido a su elevado contenido en carbonato cálcico (74,6%) ya que se generaron a altas temperaturas. El carbonato cálcico se endurece cuando entra en contacto con el agua de la lluvia y forma una costra de baja permeabilidad (Balfour y Woods, 2006; Onda et al., 2008). Esta costra se observó después de la simulación de lluvia, y en este caso, las cenizas y no el suelo, controlaron la escorrentía producida.

Durante la segunda lluvia realizada una semana después, antes de la cual el suelo se había secado de nuevo, el coeficiente de escorrentía en suelo desnudo no fue significativamente diferente al de la lluvia anterior. Respecto a las parcelas cubiertas con las cenizas recogidas en el incendio a baja severidad, éstas tuvieron una infiltración del 100% en todos los espesores, siendo mayor que el suelo desnudo. Según la hipótesis de Kinner y Moody (2010), en la primera lluvia, en condiciones secas, las cenizas actúan como una barrera capilar absorbiendo agua hasta que se saturan. Sin embargo, en condiciones húmedas o tras una lluvia, las cenizas pierden su estructura y sus propiedades capilares y por tanto, conducen el agua mejor que el suelo y reducen su capacidad de almacenamiento. Así, las cenizas que ya habían perdido su estructura (observación a simple vista), transmitieron el agua hacia el suelo más fácilmente que en la primera lluvia. Además, el suelo pudo haber perdido la ligera repelencia al agua por contacto con las cenizas y estuvo protegido por éstas de la compactación del propio suelo, ambos factores importante en la producción de escorrentía del suelo desnudo y, por tanto favoreciendo la tasa de infiltración del suelo. En el experimento anterior, estas cenizas taponaron los poros del suelo franco-arenoso y desestructurado tras la primera lluvia, pero en este caso son demasiado gruesas para taponar este suelo de textura franca y más compacto. Así, el tipo de suelo es importante para determinar si existirá bloqueo de poros por las cenizas o no. Respecto a las cenizas producidas a alta intensidad, en la segunda lluvia se reduce la escorrentía respecto a la primera lluvia sólo en la capa de 5 mm ya que el suelo ha quedado descubierto en parte en mayor proporción que las parcelas con 5 mm de espesor de cenizas recogidas en el incendio. Sin embargo, en 15 y 30 mm la escorrentía fue ligeramente menor, aunque no significativamente diferente a la primera lluvia. Los resultados indican que la costra formada por carbonato cálcico sigue haciendo efecto aunque con algunas fisuras por donde el agua penetra al suelo.

Entre las replicas realizadas en este experimento, se detectó una variabilidad mayor que entre las réplicas del experimento anterior. Esta variabilidad se debe a la naturaleza del experimento en el campo, donde algunos factores no se pueden controlar, en este caso la exacta reproducibilidad de las parcelas. Algunas parcelas presentaban macroporos formados por fauna, raíces, materia orgánica o incluso ligera repelencia al agua. Sin embargo, las tendencias para cada tipo de cenizas, grosor y evento de lluvia son claras. Las cenizas producidas a baja intensidad recogidas en el incendio aumentan levemente la escorrentía y la reducen tras la primera lluvia y, las cenizas producidas a alta intensidad aumentan la escorrentía respecto al suelo desnudo debido al encostramiento. Además, en la primera lluvia las cenizas retrasan la aparición de la escorrentía proporcionalmente a su grosor debido a su elevada capacidad de almacenar agua. De esta manera, las cenizas producen un efecto u otro en la hidrología del suelo según el tipo y el evento de lluvia, igualmente que las mismas cenizas produjeron efectos distintos, comparándolas con el experimento anterior, según el tipo de suelo. La Figura 13 muestra un resumen de los efectos de las cenizas observados en estos experimentos en contexto con los todavía escasos estudios realizados en otros lugares.

En cuando al sedimento producido, en todos los casos fue proporcional a la escorrentía producida ($r = 0,9715$, $p < 0,05$), siendo mayor en los menores grosores de cenizas y sobre todo en las cenizas de laboratorio producidas a alta temperatura. En las parcelas recubiertas con cenizas formadas a baja severidad se recogieron de media 22 g m^{-2} en 5 mm de grosor, 10 g m^{-2} y 1 g m^{-2} en 15 y 30 mm. En cambio, en las parcelas cubiertas por las cenizas quemadas a alta intensidad, se recogieron 318, 196 y 111 g m^{-2} en 5 mm, 15 y 30 mm respectivamente.

El análisis químico de estas escorrentías confirmó la hipótesis inicial sobre la existencia de diferencias en la composición química del agua procedente del lixiviado de las cenizas en el laboratorio y la cantidad de nutrientes liberados en la escorrentía. El lixiviado de las cenizas recogidas en el incendio forestal a baja severidad tuvo un pH de 9 y conductividad eléctrica de 2620 µs cm^{-1} con elevadas cantidades de Ca^{2+} (73 ppm), Na^{+} (35 ppm) y Cl^{-} (60 ppm) y menores de Mg^{2+} (15 ppm), K^{+} (12 ppm) y NO_3^{-} (1 ppm). En cambio el pH en el agua de la escorrentía descendió a 8 y la concentración de nutrientes disueltos en ésta durante la lluvia simulada fueron menores para el Ca^{2+} (30 ppm), Mg^{2+} (9 ppm) y Na^{+} (3 ppm), igual cantidad de K^{+} y Cl^{-} y mayor de NO_3^{-} (15 ppm). El lixiviado de las cenizas quemadas en el laboratorio de *Citrus sp.* a alta

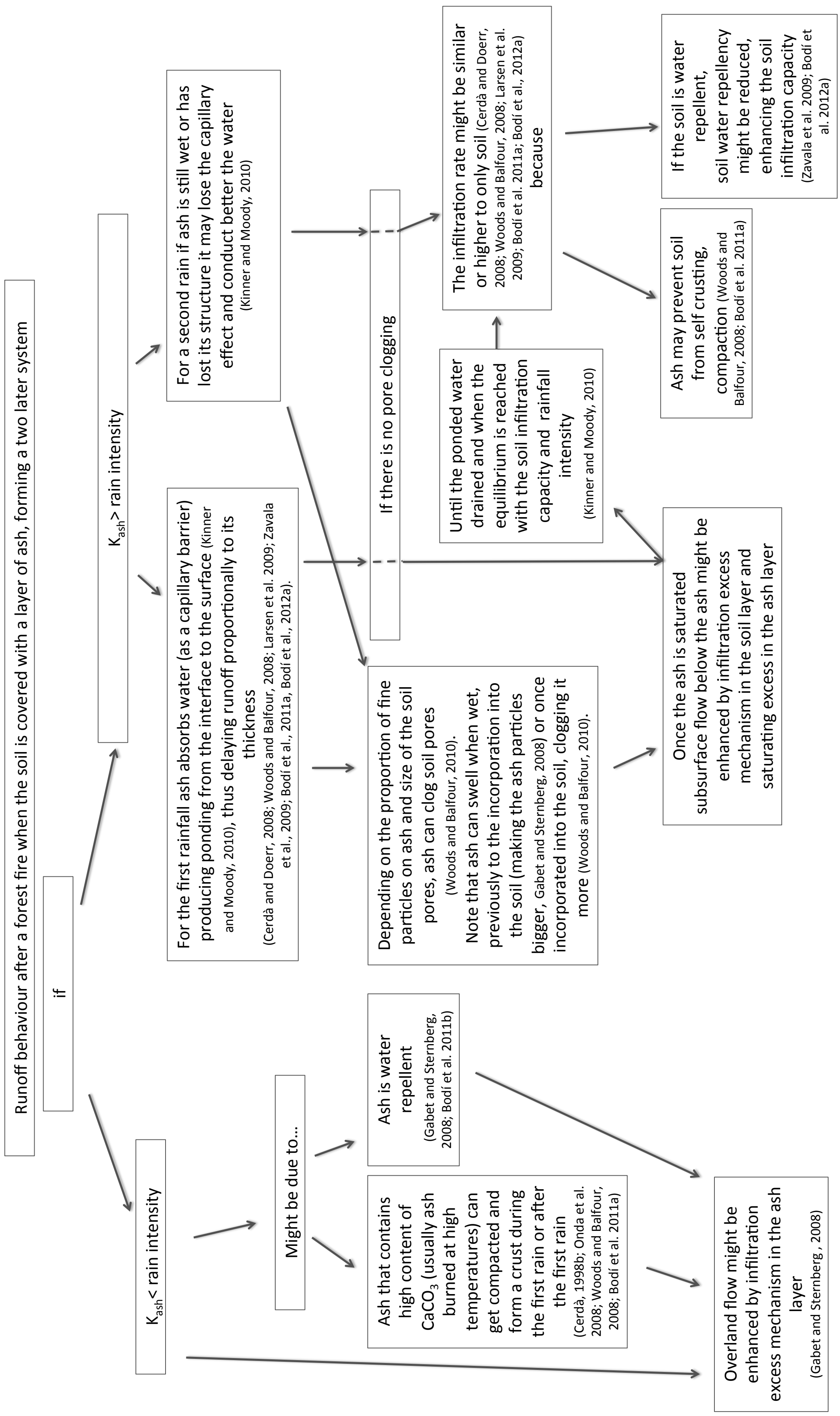

Figura 13. Factores y mecanismos por los que se produce o reduce la escorrentía producida en un sistema de dos capas compuesto por cenizas y suelo que se encuentra tras un incendio forestal.

intensidad correspondió con el de unas cenizas más mineralizadas con pH más elevado 12 y una conductividad eléctrica de 14390 µs cm^{-1}, aunque no se midió nada de Ca^{2+} y Mg^{2+} y poco NO_3^- (8 ppm) y altas cantidades de Na^+ (115 ppm), K^+ (1146 ppm) y Cl^- (296 ppm). En el agua de la escorrentía en cambio si que se incrementan los niveles de Ca^{2+} hasta 40 ppm y Mg^{2+} hasta 2 ppm y el NO_3^- también se incrementó hasta 10 ppm, pero el Na^+, K^+ y Cl^- descendieron considerablemente a 15, 195 y 71 ppm respectivamente. El suelo pareció no influir aportando más a estos niveles de nutrientes, sino puede que reteniendo parte de ellos ya que en todos los casos los valores obtenidos para cada nutriente fueron menores que en los dos tipos de cenizas. Por tanto, los motivos del descenso de los nutrientes pudieron ser causados por diferentes motivos como la retención en el suelo de parte de lo solubilizado, por lixiviación de parte de los nutrientes con el agua de infiltración o porque no hubo suficiente cantidad de agua para poder solubilizarlos todos durante la lluvia (Khanna et al., 1994). Por otra parte, el incremento en ciertos nutrientes, NO_3^- para ambas cenizas y de Mg^{2+} y especialmente Ca^{2+} en el caso de las cenizas quemadas a alta temperatura, pudo deberse a que estaban en el suelo en posiciones de cambio y se han liberado al intercambiarse con otros cationes que vienen de la disolución de las cenizas o a que el agua utilizada en la lluvia simulada pasara a ser algo ácida al mezclarse con la atmósfera y ser capaz así de disolverlos a partir de los carbonatos que son poco insolubles en agua destilada y muy elevados en estas cenizas, un 75% (Soto y Díaz-Fierros, 1993; Ulery et al., 1993).

Durante la segunda lluvia simulada, la escorrentía del suelo mantuvo valores de nutrientes similares a la primera lluvia y en las parcelas cubiertas por 5 mm de cenizas los valores de nutrientes fueron parecidos a los del suelo que quedó más descubierto. En las parcelas con los iniciales 15 y 30 mm de grosor de capa de cenizas, todos los nutrientes descendieron excepto el NO_3^- que se incrementó a 40 ppm en ambas cenizas (en el suelo es de 6 ppm) y en las cenizas quemadas a alta intensidad el Mg^{2+}. Los motivos del aumento de los NO_3^- pudieron deberse al aumento de las tasas de nitrificación por microorganismos tras esta fertilización con las cenizas, donde dos semanas después se observó mayor crecimiento de *Brachypodium retusum* (Christensen, 1973; Mataix-Solera y Guerrero, 2007; Raison et al., 2009). En cambio, el aumento del Mg^{2+} pudo deberse en parte al intercambio entre elementos solubilizados de las cenizas con elementos retenidos en el suelo o a la disolución de Mg^{2+} a partir del $MgCO_3$ tras una semana a la intemperie. Es por tanto que aunque el análisis químico de las cenizas utilizando los lixiviados es útil para saber la composición de las cenizas, estos nutrientes no son necesariamente los mismos que se liberaran durante una lluvia y serán arrastrados en el agua, provocando cambios en la calidad de ésta.

4. Conclusiones/Conclusions

4. Conclusiones

Esta investigación contribuye a comprender el papel de las cenizas como una capa dinámica y efímera que cubre la superficie del suelo tras un incendio forestal y su interacción con el sistema suelo, prestando particular atención a su influencia en la repelencia al agua del suelo. Las cenizas producen efectos edafológicos, hidrológicos y geomorfológicos en la respuesta del ecosistema tras un incendio forestal. Las mayores conclusiones de los experimentos y medidas realizados tanto en el laboratorio como en el campo son:

i) La repelencia al agua es una propiedad relativamente común en suelos calcáreos mediterráneos de bosques maduros. Los incendios forestales suelen aumentarla y en nuestro caso hemos encontrado incrementos a 1 cm de profundidad, aunque se reduce a niveles muy bajos tras un año probablemente debido al recubrimiento de la superficie del suelo con sedimento compuesto de suelo y cenizas hidrofílicas junto con la falta de nuevas deposiciones de material orgánico y hidrofóbico. El restablecimiento de la repelencia al agua a niveles de antes del incendio puede durar más de 10 años según los resultados de la investigación en la Sierra de Enguera. En esta zona de estudio la repelencia al agua variable temporalmente debido a los cambios en el contenido de humedad del suelo y espacialmente, a escala de milímetro y de cuenca, debido a las diferencias en las características de la superficie del suelo. Sin embargo, este trabajo muestra que la probabilidad de encontrar repelencia al agua en este tipo de suelos se puede predecir en bosques maduros a partir de variables simples como el tipo de vegetación, la profundidad del suelo y su contenido de humedad.

ii) Las cenizas de ciertas especies vegetales pueden ser repelentes al agua cuando se generan en incendios de baja severidad, es decir a bajas temperaturas (< 400ºC), y son por tanto resultado de un bajo grado de combustión. Temperaturas más elevadas producen cenizas hidrofílicas. La repelencia al agua está relacionada con el contenido y tipo de materia orgánica de las cenizas.

iii) Una capa de cenizas de más de 5 mm de espesor puede reducir la repelencia al agua del suelo al incrementar la presión hidráulica y el contacto entre el agua y el suelo y, por tanto promover flujos preferenciales, sobre todo durante de la primera lluvia tras el incendio. Cuando las cenizas están incorporadas en el suelo, pueden aumentar o reducir la repelencia al agua del suelo dependiendo su naturaleza hidrofílica o repelente al agua.

iv) La capa de cenizas puede reducir la escorrentía superficial proporcionalmente a su espesor debido a su elevada capacidad de almacenamiento de agua, y por tanto retrasar el inicio de la escorrentía, sobre todo durante la primera lluvia tras un incendio. Sin embargo, en algunos casos pueden favorecer el incremento de la escorrentía superficial cuando se compactan (si contienen una elevada cantidad de carbonatos) o cuando son repelentes al agua. Las cenizas también pueden modificar la capacidad de infiltración del suelo que cubren de dos maneras: (a) reduciendo su repelencia al agua o evitando su encostramiento (las cenizas actúan como un acolchado) lo que aumenta la capacidad de infiltración del suelo; o (b) taponando los poros del suelo, proceso que depende del tamaño de la partícula de las cenizas y del suelo, y que resulta en una reducción en la capacidad de infiltración del suelo.

v) Las cenizas protegen el suelo de la erosión por impacto de las gotas de lluvia en todos los casos y previenen la erosión laminar siempre que no se saturen y no ocurra escorrentía superficial. El sedimento producido en los suelos cubiertos de cenizas está asociado a la escorrentía superficial generada. Asimismo, durante episodios de lluvias intensas, las cenizas pueden aumentar la capacidad de transporte del sedimento contribuyendo a eventos extremos de erosión.

vi) Los nutrientes liberados de las cenizas pueden modificar la calidad del agua de la escorrentía, incrementando su pH, conductividad eléctrica y sobre todo el contenido de cationes. Los nutrientes liberados no son necesariamente los mismos que la composición elemental de las cenizas debido a las interacciones químicas con el agua de la lluvia y el suelo.

4. Conclusions

This research contributes to the understanding of ash as a dynamic and ephemeral layer that covers the soil surface after a wildfire and its interaction with the hidrological processes of the soil system, with particular attention to its influence on soil water repellency. Ash effects determine the pedological, hydrological and geomorphological response of the landscape after a forest fire. The main conclusions from the laboratory and field experiments carried out are:

i) Water repellency is a common property in mature Mediterranean calcareous forest soils. Forest fires usually produce an increase in the degree of soil water repellency. In the cases studied here, this increase occurred at around 1 cm of soil depth, but the water repellency in burned soils decrease to very low levels after one year. It is suggested that this was because the topsoil was covered with wettable material composed of ash and soil, together with the lack of fresh organic and hydrophobic material inputs after fire. The results obtained at the Sierra de Enguera study sites indicate that the recovery of the pre-fire water-repellent conditions may take more than 10 years. In this sites water repellency is variable temporally due to the changes in soil moisture and spatially, at millimetre and catchment scale, due to differences between surface characteristics. However, the work conducted here also shows that the probability of occurrence of water repellency in the mature forests in this type of soils can be predicted from simple variables as vegetation type, soil depth and moisture content.

ii) Ash from certain vegetation type can be water repellent when it is generated in low severity fires, i.e. at low temperatures (< 400ºC), resulting in a low degree of combustion completeness. Higher temperatures yield hydrophilic ash. Water repellency is related to the quantity and quality of organic matter on ash.

iii) A layer of ash of more than 5 mm thick can reduce soil water repellency by increasing the hydraulic pressure and the contact between water and soil, and hence promoting fingered subsurface flow, especially during the first rain event after the fire. When ash is incorporated into the soil, it can increase or reduce soil water repellency depending on its wettable or water repellent nature.

iv) An ash layer can reduce overland flow proportionally to its thickness due to its high water storage capacity, and thus delaying the onset of overland flow, especially during the first rainfall after a fire. However, in other cases, ash can favour an increase in the overland flow when it becomes compacted (if contains high levels of carbonates) or it is water repellent. Ash can also modify the infiltration rate of the underlying soil by (a) decreasing the degree of soil water repellency and by reducing the soil's susceptibility crusting (ash acts as a mulch), resulting in a higher soil infiltration rate; or (b) by enhancing pore clogging, which depends on the particle sizes of soil and ash, and results in a reduction of the soil infiltration rate.

v) Ash does protect the soil from splash erosion and will prevent sheet erosion so long as the ash layer is not saturated and overland flow does not occur. The sediment yield produced from soils covered with ash is associated with overland flow. However, during intense and\or large rainfall events ash can increase the transport capacity of the sediment, contributing to high erosion events.

vi) The nutrients released from ash can modify runoff water quality increasing its pH, electrical conductivity and especially cation content. The nutrients released are not necessarily the same as the elemental composition of ash itself due to its chemical interactions with water from rainfall and soil.

5. Referencias bibliográficas

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6. Anexos

6.1. Efectos de los incendios forestales en la vegetación y el suelo en la cuenca mediterránea: revisión bibliográfica. Boletín AGE, 58, 33-55

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EFECTOS DE LOS INCENDIOS FORESTALES EN LA VEGETACIÓN Y EL SUELO EN LA CUENCA MEDITERRÁNEA: REVISIÓN BIBLIOGRÁFICA

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RESUMEN

Los ecosistemas mediterráneos son susceptibles a los incendios forestales y han evolucionado con la recurrencia de este factor ecológico natural desde el Plioceno. Con el control del fuego por el ser humano se incrementó la frecuencia de los incendios para gestionar el entorno. Sin embargo, la actual despoblación y abandono de la montaña mediterránea europea ha provocado un aumento de la cubierta vegetal y un incremento de incendios forestales sin control y de alta intensidad. Esta revisión bibliográfica muestra que después de un incendio forestal en el Mediterráneo, el recubrimiento vegetal es relativamente rápido y que los efectos del fuego en el suelo y las tasas de erosión no suelen ser catastróficos excepto el primer año. No obstante, los efectos pueden ser muy variables según la intensidad y severidad del fuego y por este motivo no siempre es necesaria una actuación post-incendio. Se concluye que se necesita una adecuada gestión del monte en la que no se obvie el fuego como elemento clave de los ecosistemas mediterráneos.

Palabras clave: fuego, vegetación, suelo, ecosistemas mediterráneos.

ABSTRACT

A review of fire effects on vegetation and soil in the Mediterranean basin. Mediterranean ecosystems are prone to forest fires and its recurrence played an important role in the evolution of these ecosystems. In addition, fires were more frequent after humans started controlling

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them with the aim of managing the landscape. On the contrary, the current depopulation and abandonment of the European Mediterranean forests resulted in an increase of the vegetation cover leading to high intensity fires that burn out of control. This bibliographic revision shows that after a forest fire, the vegetation recovery is relatively fast and fire effects on soil and erosion rates are not catastrophic except the first year. However, the effects of fire in the ecosystems are different depending on the intensity and severity of the fire. For this reason, post-fire treatments are not always necessary. As a conclusion, an appropriate forest management that does not obviate the fire as a key element of the Mediterranean ecosystems is needed.

Key words: fire, vegetation, soil, Mediterranean ecosystems.

I. INTRODUCCIÓN

El fuego tiene un importante papel en los procesos que gestionan el Sistema Tierra. La abundancia de carbones en los sedimentos demuestra que el fuego ha actuado desde hace 400 millones de años (principios del Devónico) cambiando su frecuencia e intensidad según los niveles de oxígeno atmosférico y el clima. Como componente del sistema terrestre, el fuego ha influido en la composición de los gases de la atmósfera, el clima, la biota, las geoformas, el transporte de materiales y las tasas de sedimentación (Scott, 2000; 2009).

La configuración de ecosistemas como los bosques boreales de coníferas, praderas, sabanas y bosques mediterráneos, es debida principalmente a la acción del fuego además de por motivos climáticos (Bond *et al*. 2004, Bodí *et al*. 2008). La recurrencia de incendios propicia un ecosistema distinto del esperado según la situación climática en la que se encuentra. En estas circunstancias de incendios reiterados, las especies con algún mecanismo de resistencia al fuego perduran y desarrollan para su propia supervivencia dispositivos de reproducción y morfologías para resistir e incluso favorecer los incendios con un régimen concreto de recurrencia (Pyne, 2001). El fuego es así un factor ecológico esencial en la distribución de los biomas de la Tierra, funcionando como un gran herbívoro (Bond y Keeley, 2005). A escala humana y regional los efectos de los incendios son heterogéneos y a veces catastróficos. No obstante, incluso dentro una misma zona, según la litología, régimen de precipitación, orientación o usos del suelo, los efectos serán variados y contrastados (Neary *et al*., 1999; Kutiel, 2006).

Esta importancia y complejidad de los incendios forestales hace que su estudio sea una cuestión relevante en determinadas zonas del mundo. En Valencia, ya hace 20 años se llevó a cabo el seminario «El papel del fuego en los ecosistemas mediterráneos» organizado por Manuel Costa en la Universidad Internacional Menéndez Pelayo. En éste, se avanzaron las investigaciones más actualizadas sobre la cuestión con la presencia de prestigiosos científicos que trabajaban (y siguen trabajando) en el estudio de los incendios forestales en ecosistemas mediterráneos: P. Trabaud, Z. Naveh, A. Pons, P. Zedler, W. Oechel y W. Christensen entre otros. Además del papel de los incendios forestales en el mediterráneo, se trataron otros temas relacionados como los efectos de los incendios en el suelo, vegetación, fauna, gestión pre- y post- incendio e incluso se adelantaron problemas tan actuales como el incremento de CO_2 en la atmosfera.

En este momento, la gestión del monte con relación a los incendios forestales sigue siendo un tema de actualidad en el que todavía queda mucho por aprender. Un ejemplo es la existencia de FUEGORED, una red temática nacional sobre los efectos de los incendios forestales en el suelo (http://grupo.us.es/fuegored), subvencionada por el Ministerio de Ciencia e Innovación y de la que forman parte la mayoría de científicos que trabajan en este tema en la Península Ibérica. En este contexto organizan colaboraciones, reuniones científicas anuales y publicaciones (ver Cerdà y Mataix coord., 2009 y Cerdà y Jordán coord., 2010). El objetivo de la presente revisión intenta resumir la investigación llevada a cabo desde aquel seminario hasta ahora, tras tres reuniones de FUEGORED, sobre los efectos de los incendios forestales en la cuenca mediterránea, centrándose en los efectos en la vegetación, el suelo y los procesos hidrológicos en el este de la Península Ibérica.

II. LOS INCENDIOS FORESTALES EN LA CUENCA MEDITERRÁNEA

Los ecosistemas mediterráneos poseen un clima de transición entre el régimen templado oceánico y el tropical seco. Su característica más importante es la coincidencia de la época seca con la cálida. En verano, el Mediterráneo se encuentra bajo la influencia de los anticiclones subtropicales secos que se retiran en invierno hacia latitudes más bajas dejando paso a las borrascas atlánticas de latitudes templadas. Esta peculiaridad propicia que los ecosistemas sean susceptibles a los incendios forestales con una vegetación agostada en verano, que junto a su naturaleza inflamable favorece la aparición y expansión de los incendios (Naveh, 1991; Arianoutsou *et al*., 1993).

El fuego ha estado presente en el Mediterráneo como fenómeno natural mucho antes de que el hombre existiera, ya fueran provocados por rayos en tormentas eléctricas o debido a erupciones volcánicas (Naveh, 1975). De hecho, los incendios fueron durante el Plioceno uno de los factores que contribuyeron a configurar, junto al clima, la vegetación mediterránea (Bond y Keeley, 2005; Mataix-Solera y Guerrero, 2007). No se sabe con exactitud cuál era el régimen de incendios natural, pero debió caracterizarse por incendios menos frecuentes y áreas afectadas más grandes, ya que el fuego actuaba sobre un paisaje más homogéneo que el actual y sin barreras antrópicas como carreteras, ciudades, pastos o campos de cultivo (Naveh, 1975; Arianoutsou *et al*., 1993).

Con la aparición de las sociedades de cazadores-recolectores se utiliza el fuego para abrir claros en el bosque y crear zonas más accesibles, favorecer la producción de ciertos alimentos y propiciar mejores zonas de caza (Naveh, 1991). Pero el mayor impacto del uso del fuego se produjo durante el Neolítico. El hombre se convierte en agricultor y ganadero, y necesita amplias zonas para pastos y cultivo que consigue al deforestar mediante el fuego. Estos incendios controlados, denominados rozas, se han seguido practicando hasta los años 60 en España y en la Europa Mediterránea también con otros fines: controlar plagas y malas hierbas, fertilizar con cenizas, mejorar pastos y modificar el bosque (Naveh, 1974; Dupré, 1983; Carcaillet *et al*., 2002).

Análisis palinológicos y arqueológicos demuestran la utilización del fuego desde hace al menos 7000 años en el territorio valenciano. Antes del uso del fuego como herramienta, predominaba el bosque mediterráneo de *Quercus ilex* y *Quercus faginea*, donde el *Pinus sp*. predominaba en las laderas. Los agricultores se instalaron obviamente en las tierras más fértiles y llanas, donde se encontraban los *Quercussp.,* que además fueron especialmente diezmados por el intenso aprovechamiento que han tenido debido a su alto valor como energía calorífica. Esto benefició la expansión de *Pinus halepensis,* que aprovechó las zonas de cultivo abandonadas y que ha sido potenciado por la repoblación forestal (Carrión y Dupré, 1996; Carcaillet *et al*., 1997; Mataix-Solera y Guerrero, 2007).

Hasta mediados del siglo XX, en España y el resto de países mediterráneos se garantizaba la gestión del monte mediante un aprovechamiento sistemático,y a veces incluso abusivo, de los recursos forestales del territorio. Los vecinos lo explotaban obteniendo leña para la lumbre y cocina, madera y pasto. Además se roturaba el suelo o se quemaba la vegetación para sembrar cereales, plantar frutales u olivos. El resultado era un paisaje compartimentado y diverso con bajo riesgo de incendios, los cuales eran rápidamente sofocados por una población que vivía en el monte y para el monte. No obstante, en algunas ocasiones tambien se producian incendios incontrolados como el de la Font Roja en Alcoy (Alicante) en 1840. Sin duda, el monte mediterráneo es el resultado de la explotación y el aprovechamiento humano (Montiel Molina, 1994; Vélez Muñoz, 1999;Molinero *et al*., 2008).

TABLA 1
ESPECIES ARBÓREAS MÁS AFECTADAS POR LOS INCENDIOS FORESTALES POR COMUNIDADES AUTÓNOMAS. DECENIO 1996-2005. (*) NAVARRA DE 2000-2005

Comunidad autónoma	Especie 1	Superficie (ha)	Especie 2	Superficie (ha)
Andalucía	*Eucalyptus camaldulensis*	18372,39	*Pinus pinea*	10385,38
Aragón	*Pinus halepensis*	2015,11	*Pinus sylvestris*	1455,17
Asturias	*Eucalyptus globulus*	5874,72	*Pinus pinaster*	4108,28
Canarias	*Pinus canariensis*	10261,83	*Pinus radiata*	52,03
Cantabria	*Quercus pyrenaica*	1346,68	*Pinus sylvestris*	1181,78
Castilla La Mancha	*Pinus pinaster*	13552,91	*Pinus halepensis*	8842,34
Castilla y León	*Pinus pinaster*	18765,23	*Quercus pyrenaica*	15391,47
Cataluña	*Pinus halepensis*	12073,61	*Pinus nigra*	11257,78
Comunidad Valenciana	*Pinus halepensis*	9881,98	*Pinus nigra*	554,94
Extremadura	*Pinus pinaster*	17887,52	*Quercus ilex*	7018,03
Galicia	*Pinus pinaster*	42518,96	*Eucalyptus globulus*	21119,72
Islas Baleares	*Pinus halepensis*	1232,64	*Quercus ilex*	11,55
La Rioja	*Quercus pyrenaica*	47,48	*Pinus halepensis*	28,66
Madrid	*Pinus pinea*	1320,93	*Pinus pinaster*	493,67
Murcia	*Pinus halepensis*	661,70	*Quercus ilex*	8,76
Navarra (*)	*Fagus sylvatica*	209,38	*Pinus halepensis*	205,58
País Vasco	*Pinus radiata*	1743,54	*Eucalyptus globulus*	285,57

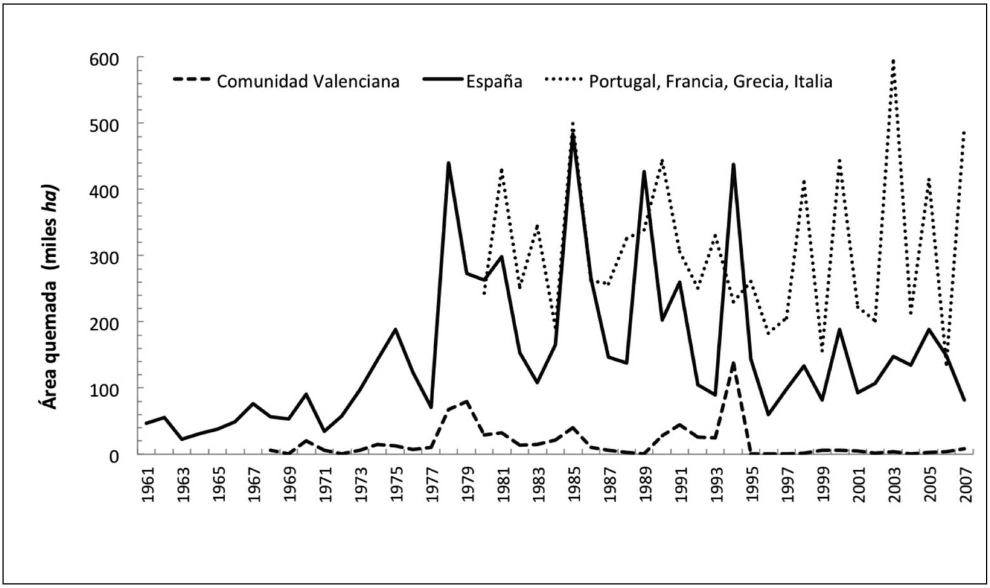
Fuente: Ministerio de Medio Ambiente, 2006

Pero en los años 60, la industrialización y el éxodo rural dieron lugar al abandono de los campos de zonas de montaña y contribuyeron a un incremento de la cubierta vegetal. En la actualidad, estas zonas no tienen apenas explotación ni se obtiene beneficio directo, ni siquiera para la gente que aún permanece en las zonas rurales, ya que nuevas normas y leyes restringen las talas, zonas de pasto y caza (Molinero *et al*., 2008). A este cambio de usos del suelo y al incremento del riesgo de incendios forestales han contribuido también las repoblaciones de pinar (la especie arbórea más afectadas por el fuego en Cataluña, Aragón, Comunidad Valenciana y Murcia es *Pinus halepensis)* (Tabla 1) y el aumento de nuevas zonas residenciales en la interfaz urbano-forestal (Pausas *et al*., 2008; Vallejo *et al*., 2009).

Las consecuencias han sido un aumento de los incendios forestales desde los años 70 en aquellos momentos en que se dan las condiciones climáticas favorables, a pesar del esfuerzo en pistas forestales, cortafuegos, infraestructuras y medios de extinción.El territorio valenciano es un ejemplo de esta relación entre abandono de las zonas interiores interior hacia el litoral y el incremento de los incendios forestales (Figura 1, 2) (Romero González, 1978; Pausas, 2004a). El fuego ocurre con mayor frecuenciay, aunque los ecosistemas mediterráneos son capaces de convivir con él, este cambio de régimen puede provocar efectos adversos. Pero los incendios forestales no son siempre eventos catastróficos (Mataix-Solera y Guerrero, 2007), losincendios de baja intensidad son de pequeño impacto y promueven la vegetación herbácea, incrementan la disponibilidad de nutrientes y aclaran los bosques, lo que promueve un hábitat más sano (Neary *et al*., 1999). Este fue el incendio controlado del matorral Mediterráneo durante milenios.

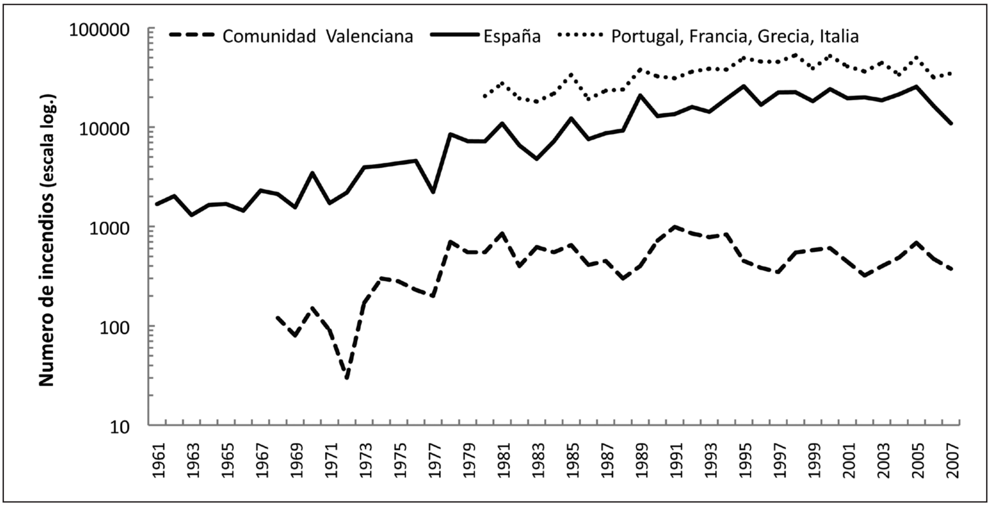
Figura 1

SUPERFICIE (MILES HA) AFECTADA POR INCENDIOS FORESTALES EN LA COMUNIDAD VALENCIANA, ESPAÑA Y OTROS PAISES DE EUROPA MEDITERRÁNEA (PORTUGAL, FRANCIA, ITALIA Y GRECIA)



Fuente: Ministerio de Medio Ambiente, 2006; European Comission, 2006; Generalitat Valenciana, 2009.

Figura 2
NÚMERO DE INCENDIOS FORESTALES ANUALES EN LA COMUNIDAD VALENCIANA, ESPAÑA Y OTROS PAÍSES DE EUROPA MEDITERRÁNEA (PORTUGAL, FRANCIA, ITALIA Y GRECIA)



Fuente: Ministerio de Medio Ambiente, 2006; European Comission, 2006; Generalitat Valenciana, 2009.

III. EFECTOS DE LOS INCENDIOS FORESTALES

El fuego modifica los ciclos biogeoquímicos, produce cambios en la vegetación, suelo, fauna, procesos hidrológicos y geomorfológicos, calidad de las aguas e incluso cambios en la composición de la atmósfera (Prodon *et al*., 1987; DeBano *et al*., 1998; Shakesby y Doerr, 2006; Moody y Martin, 2009; Raison *et al*., 2009).

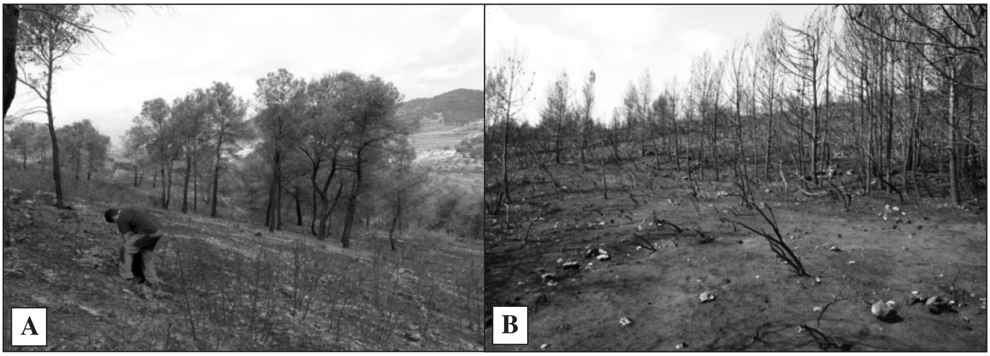
Los efectos de los incendios son muy variados debido a los múltiples factores de los que depende el incendio: biomasa disponible, intensidad (temperaturas alcanzadas y duración), área quemada, tiempo desde el último incendio, tipo de suelo, humedad, pendiente y vegetación (Neary *et al*., 1999). Así, se conforma en cada ecosistema un régimen de incendios concreto. Sin embargo, en un mismo ecosistema e incluso en un mismo incendio, la severidad, entendida como el grado de impacto en el ecosistema (Keeley, 2009), y efectos del fuego son diferentes y resultan en un mosaico de manchas de vegetación y suelo que se recuperará con o sin rehabilitación y restauración posterior (Figura 3). En esta recuperación, los efectos del fuego sobre la vegetación y los suelos son esenciales ya que influyen directamente sobre la evolución del resto del ecosistema.

1. Efectos del fuego en la vegetación

El aparente vacío biológico que queda tras un incendio es fugaz. Es frecuente observar una gran actividad biológica posterior y plantas simbiontes con algas tras las primeras lluvias (Mataix-Solera y Guerrero, 2007). Además, hay especies vegetales que presentan adaptaciones encaminadas a resistir y propagar el fuego, algunas incluso han desarrollado mecanismos reproductivos y ciclos vitales que dan ventaja a la especie cuando el fuego está presente. Todos estos signos denotan cierta compatibilidad y facilidad de recolonización (Arianoutsou *et al*., 1993).

Figura 3

INCENDIOS CON DIFERENTE SEVERIDAD INCENDIOS: A. INCENDIO EN PINOSO (ALICANTE) DONDE ALGUNOS DE LOS PINOS NO SE HAN QUEMADO COMPLETAMENTE, PERO CON MANCHAS DE CENIZAS BLANCAS QUE INDICAN LA TOTAL COMBUSTIÓN EN ALGUNAS ZONAS. B. INCENDIO EN LA SIERRA DE ENGUERA (VALENCIA), SE PUEDEN VER LOS PINOS COMPLETAMENTE QUEMADOS



Fotografía de los autores.

Existen dos grandes grupos de especies vegetales según su respuesta al fuego: las especies rebrotadoras y las no rebrotadoras o germinadoras. Las primeras tienen la capacidad de rebrotar después de un incendio y en las segundas mueren los individuos pero no las semillas. También hay especies rebrotadoras que poseen además semillas adaptadas al fuego (especies facultativas) y finalmente hay especies que no pueden rebrotar, ni sus semillas resisten las altas temperaturas. Éstas desaparecen temporalmente después de un incendio y colonizan el espacio desde los extremos de la zona quemada (Pausas, 2004b; Lloret y Zedler, 2009).

El rebrote es uno de los mejores mecanismos de piroresistencia. Para ello se desarrollan cortezas gruesas y poco inflamables, que actúan como aislantes térmicos que protegen a la planta para que después del incendio sea capaz de rebrotar. En estos casos, las partes aéreas se queman pero se mantiene viva la cepa, que en algunos casos está constituida por un tejido llamado *lignotuber* que actúa como almacén de agua y nutrientes para asegurar la pervivencia de la planta (Molinas y Verdaguer, 1993). Cada rebrotadora dispone de distintas posibilidades de resistencia según la severidad del incendioy el momento de crecimiento de la planta. Los ejemplos son múltiples en la cuenca mediterránea: *Quercus suber*, con una magnifica corteza aislante (el corcho), *Quercus coccifera, Quercus ilex, Juniperus Oxycedrus, Erica multiflora, Chamaerops humillis*, así como en el chaparral californiano, el «fynbos» de Sudáfrica y los bosques australianos de «jarrah» (Arianoutsou *et al*., 1993)

Quercus coccifera es la especie protagonista durante la recuperación de la cubierta vegetal forestal en la cuenca mediterránea. Esta especie tiene una gran capacidad de rebrote inmediatamente después del incendio y un gran sistema radicular que permite en 2 ó 3 años un

90% de recubrimiento (Sala *et al*., 1990). Trabaud (1973) experimentó quemando esta especie reiterativamente con diferentes frecuencias (cada 6, 3 y 2 años) y en diferente estación durante 19 años y *Quercus coccifera* no tuvo ningún síntoma de degradación junto con *Rubia peregrina y Brachypodium ramosum*. No obstante, otros estudios apuntan que, en áreas más áridas al sur, *Quercus coccifera* puede tener un decrecimiento tras alternancias repetitivas (Delitti *et al*., 2005). Otras especies incrementan su población después de los incendios: *Lonicera implexa* (Trabaud, 1973), *Phyllirea angustifolia, Erica arbórea* (Moreno *et al*., 2004), plantas perennes como *Ampelodesmos mauritanica* (Lloret *et al*., 2003)y geófitas dotadas de bulbos *Gladiolus, Asphodelus, Ophrys, Acis nicaeensis* (Naveh, 1975; Diadema *et al*., 2007). Pero por otra parte, hay especies que tienen tendencia a decrecer con el aumento de la recurrencia de incendios: *Erica multiflora* (Lloret *et al*., 2003), *Rhammus alaternus y Smilax aspera* (Trabaud, 1973). Sin embargo, en general parece ser que hay diferencias regionales en su habilidad rebrotadora (Pausas *et al*., 2008) (Figura 4, A, B, C, D).

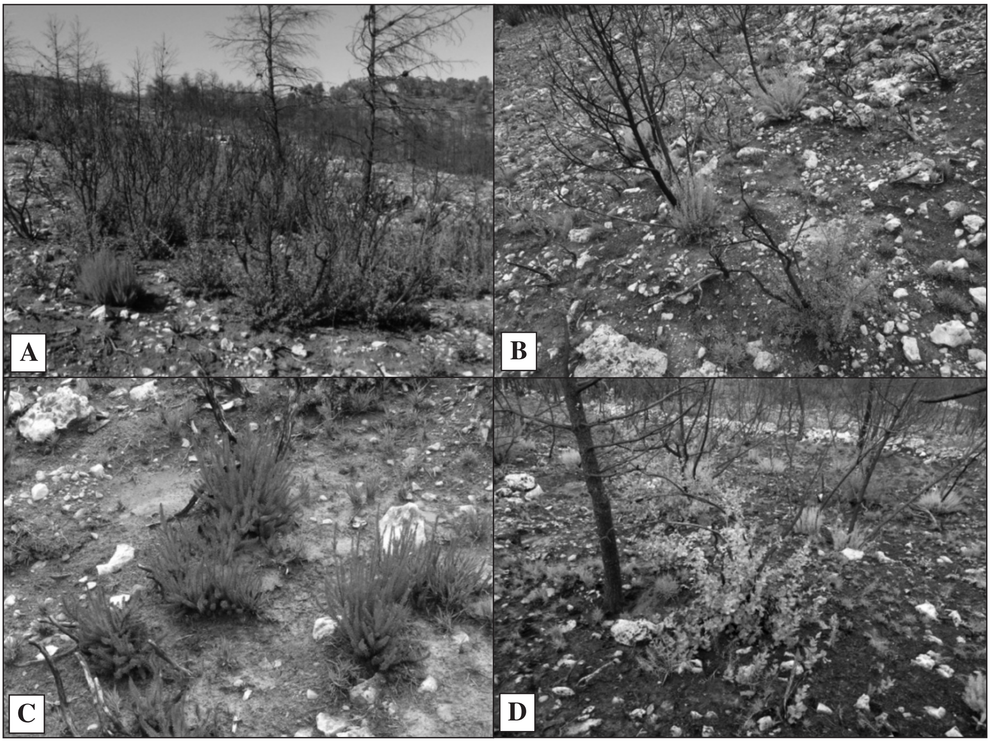
Por otra parte, las plantas germinadoras adaptadas a los incendios retienen las semillas por largo tiempo hasta que son estimuladas por el calor y se dispersan. Son semillas serótinas. En estos casos, los individuos no resisten al incendio y son substituidos por otros que nacen de sus semillas y que encuentran un espacio sin competencia, donde llega mucha luz y el suelo es rico en nutrientes (DeBano *et al*., 1998; Pausas, 2004b). Buenos ejemplos son las piñas que se abren con el calor y dispersan los piñones, en especies como *Pinus halepensis, Pinus brutia, Pinus pinaster,* permitiendo una rápida regeneración del pinar (Arianoutsou *et al*., 1993). Otros pinos con piñas no serótinas como el *Pinus pinea*, tienen piñones muy duros que resisten a los incendios. Pero hay otras especies de pinos muy sensibles (*Pinus nigra, Pinus sylvestris*) que pueden ser eliminados temporalmente según la extensión del incendio (Pausas *et al., 2008)*. Existen arbustos no rebrotadores que tienen bancos de semillas persistentes en el suelo y resisten el calor del fuego, como las *Cistus sp*. o *Ulex parviflorus* (que además acumula mucha necromasa en su parte aérea y cuando llega el fuego es altamente combustible). Hay otras especies en que las semillas son estimuladas para germinar por otros productos derivados del incendio como el humo o las cenizas (por ejemplo: *Rhamnus alaternus, Alnus glutinosa, Cistus incanus, Clematis vitalba*) (Crosti *et al*., 2006; Paula *et al*., 2006; Reyes y Casal, 2006). Sin embargo, si el periodo entre incendios no es suficiente para que los individuos lleguen a la edad adulta, o si este intervalo corto entre incendios es muy recurrente, el banco de semillas se puede agotar (Pausas, 2004b).

Después de un incendio no hay sucesión en el sentido de que unas comunidades reemplazan a otras, sino que hay una progresiva reaparición de las especies que pertenecían ya a la comunidad (Trabaud, 1990). Tras el incendio, las herbáceas de vida corta cubren el suelo junto con los arbustos y leñosas que rebrotan rápidamente. Las herbáceas alcanzan su pico en 1-5 años, y luego reducen su presencia y diversidad. Hay algunas para las que el fuego es esencial, ya que sólo aparecen tras el incendio al estar sus semillas en estado latente (De Lillis y Testi, 1990). Los rebrotes de leñosas son siempre muy rápidos debido al sistema raticular bien desarrollado del que disponen, lo que permite captar agua y nutrientes, aunque también las no rebrotadoras como *Cistus sp., Ulex parviflorus y Rosmarinus officinalis* aparecen pronto. El crecimiento es muy activo en el primer quinquenio y luego se ralentiza hasta ser casi nulo 25 ó 30 años después del incendio (Ferran y Vallejo, 1992; Arianoutsou *et al*., 1993; Ferran y Vallejo, 1998). En este aspecto, las variaciones son grandes y dependen de las características del incendio y del lugar donde se produzca (Tabla 2). Esta rapidísima regeneración es muy importante para proteger el suelo de la erosión.

Se ha comprobado que las comunidades vegetales de los ecosistemas mediterráneos tienen en general una alta resiliencia a los incendios forestales, es decir tienen gran habilidad para volver a las condiciones anteriores a la alteración (Lloret y Zedler, 2009). La vegetación resistente al fuego ha evolucionado hasta conseguir un mantenimiento y regeneración con un determinado régimen de incendios. Cuando un régimen de incendios permanece en un ecosistema por largo tiempo se produce un tipo de relación entre la resiliencia de las especies y el régimen de incendios (Lloret y Zedler, 2009). Según Naveh (1990), para estas especies el fuego puede que sea la única manera de asegurar el rejuvenecimiento. Pero no hay que olvidar que las especies están adaptadas a un régimen de incendios, no a cualquier incendio, y el cambio de régimen de incendios puede tener impactos importantes en la sostenibilidad de algunos de los componentes (Pausas y Keeley, 2009).

Figura 4

ESPECIES REBROTADORAS 6 MESES DESPUÉS DE UN INCENDIO EN LA SIERRA DE ENGUERA (VALENCIA), A FINALES DE ABRIL DE 2008. A. QUERCUS COCCIFERA, UNA DE LAS PRIMERAS EN REBROTAR. B. JUNIPERUS OXYCEDRUS Y RHAMNUS LYCIOIDES.C. ERICA MULTIFLORA. D. QUERCUS ILEX



Fotografía de los autores.

Tabla 2

CUBIERTA VEGETAL (%) EN SUELOS MEDITERRÁNEOS DESPUÉS DE UN INCENDIO (T, TIEMPO EN MESES)

Autor y año	**Cubierta vegetal (%)**	**Características**	**t (m)**	**Vegetación antes del incendio**	**Zona de estudio**
Cerdà (1998b)	20 116		12 72	Bosque de pinos y matorral denso	Los bosques, Pedralba, Valencia, España
Cerdà y Doerr (2005)	55,6 86,4 125,9	Alta severidad	12 24 72	Bosque de pino y matorral denso	Serra Grossa, Valencia, España
Cerdà y Lasanta (2005)	50		4	Matorral	Valle de Aisa, Pirineos, España
De Lillis y Testi (1990)	20 18 3	*Myrtus communis* *Spartium junceum* *Brachypodium ramosum*	12	Matorral	Costamezza, centro de Italia
Gimeno-García *et al*. (2007)	26 19 29 9 12 45 16 42,5	Control Altaseveridad Baja severidad Alta severidad Bajaseveridad Control Alta severidad Baja severidad	0 12 96	Matorral	La concordia estación experimental, Lliria, España
Inbar *et al*. (1998)	10-30 50-70		12 24	Bosque de pino y matorral mediterráneo	Mont Carmel, Israel
Marqués y Mora (1992)	50 25	Norte Sur	15	Bosque de pinos y matorral de Rosmarinus-Ericion al Norte y de *Quercus coccifera* al Sur	Montserrat, Cataluña, España

Autor y año	Cubierta vegetal (%)	Características	t (m)	Vegetación antes del incendio	Zona de estudio
Mayor *et al*. (2007)	50 78 97	Alta severidad Control	48 72 72	Bosque de pinos y matorral	Montaña de Xortà, Alicante, España. Orientación sur. Terrazas abandonadas
	7138	Calizas			
	72,85	Calizas			
	40,45	Margas			
	54,14	Margas	10		
	53,63	Termomediterráneo	34		Diferentes lugares de la Comunidad Valenciana
	62,10	Temomediterráneo	10		
	58,20	Mesomediterráneo	34		
	62,74	Mesomediterráneo	10	Bosque de pinos y matorral	
Pausas *et al*. (1999)	19,51	Herbáceas Norte (N)	34		
	39,55	Árboles-arbustos N	10		
	10,02	Herbáceas Sur (S)	34		
	24,36	Árboles arbustos S			Requena
	42,55	Rebrotadoras N	12		
	18,37	No-rebrotadoras N			
	22,01	Rebrotadoras S			
	10,85	No rebrotadoras S			
Trabaud (1990)	50	Herbáceas	12	*Quercus coccifera* garriga	Montpellier, Francia

2. Efectos del fuego en el suelo

El suelo es el componente básico del ecosistema forestal. Su sostenibilidad y recuperación dependen de las funciones y procesos químicos, físicos y biológicos que ocurran debajo de la capa de hojarasca (Neary *et al*., 1999; Mataix-Solera y Guerrero, 2007). Tras el fuego, el suelo puede sufrir cambios directos producidos por el calentamiento y la combustión, e indirectos como consecuencia de la situación microclimática después de la pérdida de la cubierta vegetal y recubrimiento de las cenizas (Figura 5, A). Estos cambios dependerán principalmente de la temperatura alcanzada durante el incendio (Neary *et al*., 1999).

El calentamiento del suelo produce variaciones en algunas de las propiedades físicas y químicas. El pH y la conductividad eléctrica normalmente aumentan, debido al aporte de carbonatos, cationes básicos y óxidos procedentes de las cenizas. El tiempo de recuperación del pH inicial es variado y se considera que es más o menos rápido según el tiempo que las cenizas permanezcan en el suelo (Mataix-Solera y Guerrero, 2007). Este aporte de cenizas también enriquece el suelo con un aumento de nutrientes (Ca, Mg, K, Na, P) y según Kutiel y Naveh (1987b) es considerado el mayor factor de crecimiento de la vegetación en los ecosistemas mediterráneos. Sin embargo hay algunos nutrientes que se pierden con el humo del fuego, se volatilizan (Raison *et al*., 1984), o existe peligro de que la acción del viento, erosión o lixiviación laven esta inyección de nutrientes fundamentales, sobre todo cuando no hay vegetación (Arianoutsou *et al*., 1993; Neary *et al*., 1999; Cerdà y Bodí, 2007). Así, esta fertilización puede ser efímera, durar 4-5 meses (Kutiel y Naveh, 1990; Gimeno-García *et al*., 2000), hasta 14 meses (Kutiel y Naveh, 1987b) o incluso 7 años (Úbeda *et al*., 2005).

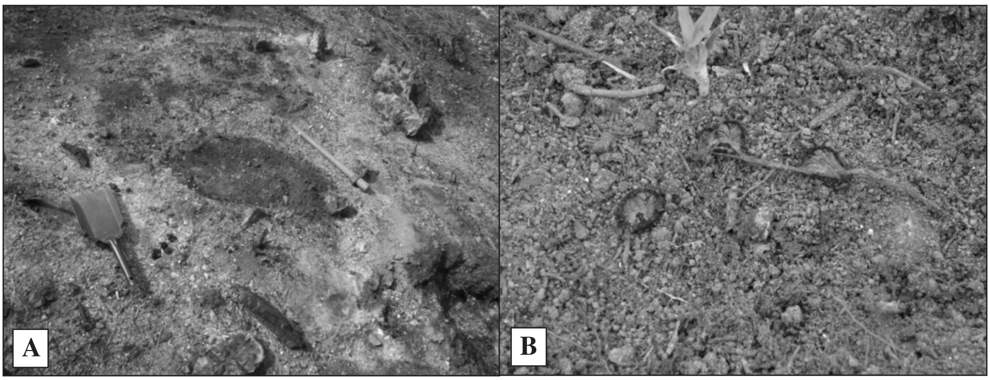
El nitrógeno es uno de los elementos que más fácilmente se volatiliza. Durante la combustión se pueden perder grandes cantidades, pero afortunadamente para los ecosistemas mediterráneos y regiones semiáridas donde es limitado, se suele encontrar tras el incendio más nitrógeno disponible en el suelo en la forma de amonio (NH_4^+-N) (Kutiel y Naveh, 1987a; Giovannini *et al*., 1990b; Gimeno-García *et al*., 2000). Asimismo, en los meses posteriores se produce un aumento del nitrógeno (N_2) por microorganismos promovidos por el incendio y especies de leguminosas fijadoras como el *Ulex parviflorus* (Arianoutsou *et al*., 1993; Pastor-López y Martin-Martin, 1995; Neary *et al*., 1999; Raison *et al*., 2009).

Respecto a los cambios en el carbono orgánico del suelo, los resultados son complejos y variados según la intensidad del incendio. En incendios de baja intensidad puede haber incrementos de carbono orgánico procedente de la vegetación parcialmente pirolizada, en cambio, a intensidades elevadas la cantidad de materia orgánica de la superficie del suelo puede disminuir (Mataix-Solera *et al.,* 2002). Según Knoepp *et al*. (2005) calentando el suelo a 450 °C durante dos horas o a 500 °C media hora, se destruye el 99% de la materia orgánica. Pero el fuego no sólo modifica la cantidad de la materia orgánica, también altera su calidad. Actúa como un agente que acelera las tasas de mineralización del carbono orgánico y además modifica las tasas de descomposición postincendio ya que, a medida que se incrementa la temperatura, el humus sufre modificaciones que le hacen más resistentes a la degradación microbiana (González-Vila *et al.,* 2009; Knoepp *et al*., 2005). Esta materia orgánica carbonizada que se produce en grandes cantidades y se acumula en el suelo, puede contribuir en un 30-40% al carbono del suelo en ecosistemas propensos a incendios forestales y al secuestro de carbono a largo plazo, siendo un componente significativo en el ciclo global del carbono (Forbes *et al*., 2006).

La estabilidad de los agregados también puede cambiar tras el paso del fuego. Generalmente se detecta una reducción de la estabilidad de los agregados relacionado con la pérdida de materia orgánica (Cerdà, 1993; DeBano *et al*., 1998; Badia y Martí, 2003). No obstante, es posible encontrar tendencias contrarias cuando hay incendios de baja intensidad en los que aumenta la materia orgánica (Díaz-Fierros *et al*., 1987), debido a fusiones de arcillas por calcinación que endurecen los agregados (Giovannini *et al*., 1990a) o incluso debido la cementación por las sustancias hidrofóbicas que los hacen más resistentes (Giovannini y Lucchesi, 1983). La porosidad y la capacidad de retención hídrica también pueden verse disminuidas al cambiar la estructura del suelo y desaparecer la materia orgánica si las intensidades son más elevadas (Neary *et al*., 1999). Estos tres factores, junto con la hidrofobicidad, son aspectos fundamentales que determinaran la aireación, infiltración y erodibilidad de un suelo, sobre todo cuando se elimina la vegetación y hojarasca.

Figura 5

A. CAPA DE CENIZAS QUE CUBRE EL SUELO DESPUÉS DE UN INCENDIO EN PINOSO (ALICANTE) EN JULIO DE 2009. B. HIDROFOBICIDAD EN UN SUELO QUEMADO EN LA SIERRA DE ENGUERA (VALENCIA) EN 2008. EL MÉTODO UTILIZADO ES ELTEST DE PENETRACIÓN DE LA GOTA DE AGUA (WDPT) QUE CONSISTE EN APLICAR GOTAS DE AGUA DESTILADA SOBRE EL SUELO Y MIDIENDO EL TIEMPO QUE TARDAN EN INFILTRARSE



Fotografía de los autores.

La hidrofobicidad o repelencia al agua puede verse provocada, aumentada o disminuida según la temperatura alcanzada en el suelo. A grandes rasgos, si la temperatura alcanzada en el suelo es de 200-250°C, la provoca o la aumenta, y si es mayor de 300°C, la destruye (DeBano, 1981). Ésta se debe a la existencia de sustancias orgánicas que se volatilizan durante la combustión y se condensan posteriormente (Doerr *et al*., 2000). Los suelos ácidos y de textura arenosa son más propensos a la hidrofobicidad (Mataix-Solera y Doerr, 2004), pero ésta también aparece en suelos calcáreos (Figura 5, B) (Cammeraat y Imeson, 1999; Cerdà y Doerr, 2005; Arcenegui *et al*., 2007). No obstante, se ha comprobado que las mismas propiedades del suelo pueden controlar si el suelo desarrolla hidrofobicidad como consecuencia de la combustión. Mataix-Solera *et al*. (2008) demostró que la baja ratio materia orgánica/arcilla y una presencia elevada de caolinita en el suelo denominado comúnmente como *terra rossa* (principalmente Rhodoxeralfs), lo hacen muy poco susceptible a desarrollar esta propiedad. En los demás suelos mediterráneos calcáreos, los factores que principalmente controlan la hidrofobicidad son, junto con la temperatura alcanzada, el tipo de vegetación y la cantidad de hojarasca presente, siendo *Pinus halepensis* y *Rosmarinus officinali*s los mayores inductores de sustancias hidrofóbicas en el suelo (Arcenegui *et al*., 2007; Tessler *et al*., 2008). Sin embargo, la repelencia al agua ocurre también en suelos no afectados por incendios forestales y depende principalmente del tipo de vegetación presente y de la humedad del suelo (Mataix-Solera *et al*., 2007; Doerr y Shakesby, 2009). La persistencia y la distribución espacial de la hidrofobicidad es un factor clave en la dinámica de las escorrentías en los suelos quemados. Como consecuencia inmediata, se produce una reducción de la infiltración y un aumento de escorrentía y la erosión, además de disminuir la humedad del suelo en la zona hidrófoba (Doerr *et al*., 2009).

Los efectos indirectos del fuego sobre los suelos se producen a partir de la desaparición de la cubierta vegetal, la adición de cenizas y el ennegrecimiento del suelo. Estas modificaciones suponen cambios microclimáticos en la humedad edáfica, temperatura y radiación solar (Raison *et al.,* 2009) que afectarán a la recuperación del sistema tanto en beneficio (menos competencia, más luz, más nutrientes) como en detrimento (mayor erosión, menos infiltración) del ecosistema (Neary *et al*., 1999).

La hidrología del suelo se modifica como consecuencia del aumento de la hidrofobicidad, reducción de la materia orgánica, disminución de la porosidad y estabilidad de los agregados, el sellado del suelo por partículas minerales o cenizas y sobre todo la reducción de la cubierta vegetal (MacDonald *et al*., 2008).

Figura 6

MOVIMIENTO DE SEDIMENTOS EN UNA CUENCA DE LA SIERRA DE ENGUERA (VALENCIA) TRAS UNA TORMENTA DE 15 MM EN 10 MINUTOS DOS MESES Y MEDIO DESPUÉS DEL INCENDIO

Fotografía de los autores.

Inmediatamente después del incendio las tasas de escorrentía son prácticamente nulas, debido a la capa de cenizas que cubre el suelo (Cerdà, 1998b; Cerdà y Doerr, 2008). Más tarde, tras la pérdida de las cenizas o su encostramiento, se produce un aumento sustancial de las tasas de escorrentía y erosión que se van reduciendo con la recuperación de la vegetación.

No obstante, las magnitudes tanto de las tasas de escorrentía como de erosión son variadas (ver revisión de Cerdà y Bodí, 2007) y las tasas de erosión no suelen ser superiores a 1 Mg ha^{-1} $año^{-1}$ y por tanto se encuentran dentro del rango de sostenibilidad y tolerancia (Cerdà y Bodí, 2007). Las tormentas individuales de gran intensidad son las que desencadenan los episodios extraordinarios de arroyada post-incendio (Figura 6). Durante estas tormentas, las cuencas quemadas generalmente responden más rápido a la lluvia que antes del incendio o que cuencas no quemadas. En las zonas mediterráneas tras el incendio de verano, el suelo está desprovisto de vegetación y queda más expuesto al poder erosivo de las lluvias torrenciales otoñales (Marqués y Mora, 1992; Shakesby y Doerr, 2006; Mayor, 2007).

La recuperación de las tasas de erosión previas al incendio es variable y en ocasiones contrastada (ver Cerdà y Lasanta, 2005). En diversos trabajos se ha comprobado que las tasas de escorrentía y erosión se recuperan a los 2 años (Marqués y Mora, 1992; Cerdà, 1998a; Cerdà y Doerr, 2005), aunque los casos habría que matizarlos. Para Cerdà (1998b), las tasas de escorrentía se igualan con las iniciales después de 4 años para las condiciones de invierno, pero 2 en verano debido a las mayores tasas de infiltración con suelos secos. En zonas más áridas, la recuperación puede durar 3-4 años (May, 1990), 6 años (Mayor *et al*., 2007) u 8, si la vegetación no se regenera rápidamente (Gimeno-García *et al*., 2007). Incluso a microescala, en zonas abruptas con piedras y vegetación quemada, la vuelta a la situación de antes del incendio se produce al cabo de un año a diferencia de las zonas llanas, donde sigue siendo alta porque hay más suelo que erosionar (Lavee *et al*., 1995). La regeneración de la vegetación se considera el factor más importante para reducir la erosión en el mediterráneo (Dieckmann *et al*., 1992; Cerdà, 1998a; Inbar *et al*., 1998).

Está claro que a la gran variabilidad espacial en los efectos de los incendios forestales se ha de sumar la variabilidad climática, orográfica, litológica y de usos del suelo que provocan que las respuestas a los incendios sean variadas (Cerdà y Bodí, 2007).

IV. IMPLICACIONES PARA LA GESTIÓN

Como se ha comprobado a través de esta revisión bibliográfica, los ecosistemas mediterráneos son dependientes de los incendios en su debido régimen de recurrencia e intensidad (Bond *et al*., 2004). Aunque el fuego es una alteración, el ecosistema está preparado para volver a la situación inicial anterior al incendio. No obstante, la política forestal tradicional en la cuenca mediterránea ha replantado durante muchos años bosques monoespecíficos de pino después de los incendios, siendo algunos muy inflamables (Pausas *et al*., 2008). En otras ocasiones hahabido intentos de introducir semillas de *Quercus sp.,* que normalmente tienen alta mortalidad (Beyers, 2009). Además, estas plantaciones se realizan mediante maquinaria pesada que produce una alteración dramática de los suelos y la vegetación. Así, la decisión de dónde y cuándo usar tratamientos de remediación después de un incendio requiere una evaluación de la severidad de incendio, el clima, suelos, topografía e hidrología de la cuenca (Robichaud, 2009). Las actuaciones post-incendio que se utilizan y están siendo estudiadas principalmente son el acolchado con restos vegetales e hidrosiembras, repoblaciones con diferentes tipos de semillas y barreras de erosión. No obstante, estos tratamientos post-incendio son muy caros y solo deben ser aplicados si el riesgo de degradación del suelo y vegetación es elevado. Por ello, es justificable y efectiva la no actuación post-incendio, tanto por razones económicas como medioambientales. Un buen ejemplo de ello lo encontramos en los casos en los que las hojas caídas de los pinos después de un incendio actúan como acolchado natural y la propia regeneración de la vegetación es suficiente para reducir las tasas de erosión (Cerdà y Robichaud coord., 2009). En caso de realizar una repoblación, la ejecución de ésta puede provocar más daño al suelo que el propio incendio, y obviamente es más caro que dejar que la vegetación se recupere por ella misma. Es necesario por tanto estudiar y analizar cada caso concreto para poder decidir si es mejor o no actuar.

Hasta ahora, las políticas forestales han optado por una estricta supresión de los incendios forestales y divulgan sólo sus efectos negativos, de manera que la sociedad ha llegado a considerar que el fuego debe suprimirse completamente. Pero, como se ha comprobado, esta política propicia los incendios de alta intensidad, precisamente los que desencadenan procesos de degradación más intensos. Ejemplos son las olas de grandes incendios como los de 1979 y 1994 en la Comunidad Valenciana y, más recientemente las de Galicia en 2006, Grecia en 2007 o el sureste de Australia en Febrero de 2009, lo que demuestra que no podemos excluir el fuego de unos ecosistemas que han convivido con él de manera natural y bajo la gestión del hombre. Es por tanto necesaria una gestión del monte que puede incluir desbroces o incluso en la que se contemple la posibilidad de usar el fuego controlado para combatir los incendios, opción que ya se aplica en Canarias y Cataluña y en países como EEUU o Australia. Además, esta fue la estrategia usada en el Mediterráneo durante milenios para abrir claros, evitar incendios no controlables, o producir pastos.

V. CONCLUSIONES

El fuego es un factor ecológico natural en los ecosistemas mediterráneos y ha contribuido a modelar el paisaje que conocemos, no solo por su recurrencia natural sino también por el uso que el ser humano ha hecho de éste. La investigación científica de los últimos 20 años ha corroborado esta visión, por lo que se acepta que el fuego es necesario para un correcto funcionamiento del ecosistema en los bosques mediterráneos. Sin embargo, el régimen de incendios ha cambiado debido a las nuevas tendencias sociales, económicas y por las políticas forestales de supresión del fuego. El reto por lo tanto es gestionar un régimen de incendios adecuado para las demandas sociales presentes y futuras evitando el riesgo que suponen los incendios y a la vez conseguir la sostenibilidad del ecosistema.

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Spatial and temporal variations of water repellency and probability of its occurrence in calcareous Mediterranean rangeland soils affected by fires

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ABSTRACT

Water repellency (WR) is a common soil property in many fire-affected ecosystems, but it also occurs in long-unburned terrain. It can vary in space at different scales (between point and pedon or slope and catchment) and time (during the same day, between seasons or years, or with a post-fire recovery period). This paper: i) reports on the occurrence and persistence of WR in fire-affected calcareous forest soils under Mediterranean climatic conditions, examining its spatial variability at macro-, meso- and micro-scales, and monthly changes with soil moisture content; and ii) develops exploratory models to estimate the probability of the natural background (not fire-induced) WR to occur through a Mixed-Effect Logistic Regression Model. Four sites with comparable soil and vegetation types were studied, all of them burned in 1979, the third again 1999, and the fourth in April 2008. All sites were sampled immediately after the fire of 2008, a further 17 times until July 2009, and once more in August 2011. At each site, 5 random plots (10 cm × 10 cm) were selected for each of the vegetation types: *Pinus halepensis, Quercus coccifera, Rosmarinus officinalis* and bare soil. In each plot we carried out 10 Water Drop Penetration Time measurements at the surface and at 1 cm depth. WR was detected in samples from all sites, although WR was most frequent at the soil surface at the sites last burnt in 1979. The recently burned site had similar WR to the long-unburned ones in surface but greater at 1 cm depth, although for both depths WR was reduced by the following year. WR was still very low in the site burned 10 years before. Variability of WR between different vegetation types was as high as within the same type at the same site, and similarly high at within sites (1 ha) and at 10 cm × 10 cm plot-scales. The lowest variability was found in bare soil plots because they were mostly wettable. An exploratory model to estimate the probability of WR occurrence was derived for each unburned site. The most powerful explanatory variables for the probability of WR to occur ($P<0.0001$) for both unburned sites were the covariate moisture content together with the category soil surface (associated to the fixed factor soil depth) for one site and *P. halepensis* (associated to vegetation type) for the other. The model input parameters are straightforward to obtain and the model may be a useful tool in estimating occurrence and fluctuation of soil WR.

1. Introduction

Forest fires are an important ecological agent in the Mediterranean (Naveh, 1975; Pausas et al., 2008), which produces sudden changes in the ecosystem (DeBano et al., 1998), including the soil system and its chemical and physical properties (Certini, 2005; Neary et al., 1999). Fire effects on Mediterranean soils have been the focus of numerous studies and various aspects of these have been evaluated in a series of reviews including Sala and Rubio (1994), Cerdà and Mataix-Solera (2009), Shakesby (2011) and Mataix-Solera et al. (2011). One of the soil physical properties that can be modified by fire is water repellency (WR), which reduces the affinity of soils for water and is produced when organic molecules with hydrophobic properties are present on soil mineral surfaces or within soil pores (Doerr et al., 2009a).

WR has been documented under a wide range of vegetation types and climates (Doerr et al., 2000). Initially, however, it had been thought not to develop in calcareous or alkaline soils ($pH>7$) (Roberts and Carbon, 1971). The first studies reporting WR in calcareous forest soils were Holzhey (1969) from burned soils under chaparral vegetation in California, Dekker and Jungerius (1990) in unburned dune sands in Netherlands, and Cammeraat and Imeson (1999) for unburned soils covered with *Stipa tenacissima* tussock in south-eastern Spain. One of

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the studies to quantify the occurrence of WR in calcareous soil under pine forest, despite their fire prone nature, was that of Mataix-Solera and Doerr (2004), who recorded water repellency of lower persistence when compared with acidic soils under pine. In addition, since then, it has been established that Mediterranean *terra rossa* soils, which developed from calcareous bedrock, have a very low susceptibility to become water repellent through burning (Mataix-Solera et al., 2008, in press).

Some studies of WR in the Mediterranean calcareous forest soil have shown that WR is highly variable in space and time, as has been shown for other environments elsewhere (Doerr et al., 2000). One of the agents of change and variability is fire. During fires, temperatures in the soil between 270 and 400 °C (depending on heating duration) can induce or increase WR, but higher temperatures can destroy it (Arcenegui et al., 2008; Doerr et al., 2009b; Robichaud, 2000). Where WR has been induced or enhanced by fire, it will typically disappear or return to pre-fire levels with time, known as "natural background WR" (Doerr et al., 2009a). Indeed, in many conifer forest ecosystems, including those with alkaline soils, WR is natural and variable property irrespective of fire, with burning adding further spatial and temporal variability (Doerr et al., 2009b; Mataix-Solera et al., 2007; Tessler et al., 2008). Where WR has been measured in alkaline Mediterranean soils, which included burned and unburned soils under a variety of vegetation types, the more frequent occurrence and WR persistence has been reported under *Pinus halepensis*, with lower values from under *Quercus coccifera* and *Rosmarinus officinalis* (Arcenegui et al., 2007; Cerdà and Doerr, 2007; Mataix-Solera et al., 2007).

WR can also change rapidly or seasonally in response to variations in soil moisture (Buczko et al., 2005; Crockford et al., 1991; Hubbert and Oriol, 2005; Keizer et al., 2008; Leighton-Boyce et al., 2005; Stoof et al., 2011). Both burned and unburned soils become less repellent or completely lose their WR when certain soil moisture content is reached, but once a water repellent soil dries out, the soil water repellency can be re-established (Doerr and Thomas, 2000). This range has been defined as "transition zone", between the water content range demarcated by the upper water content, below which samples are water repellent, and the lower water content, above which samples are wettable (Dekker et al., 2001). The role of moisture in driving the temporal dynamics of WR has been examined in detail for acidic Mediterranean soils under *Eucalyptus* forest by Leighton-Boyce et al. (2005). To the authors' knowledge, however, no studies on the variations of water repellency in response to changes in soil moisture have been carried out for the very widespread calcareous Mediterranean forest soils.

In terms of the environmental implications of soil WR, which include reduced infiltration capacity, increased runoff rates and preferential flow, and decreased nutrient and water availability for plants (Doerr et al., 2000), determining the parameters that control its variability and predicting soil WR is essential for the effective management of the forest and its soils, and for flood control. Several studies have explored the relationship of WR with organic matter, pH, soil moisture and soil texture (Doerr et al., 2006; Keizer et al., 2008; Martínez-Zavala and Jordan-Lopez, 2009; Mataix-Solera et al., 2007). The relationships obtained, however, have not always been consistent. The potentially influential parameters (texture, organic matter, specific water content) seem to be poor general predictors of WR, whereas land use or vegetation and the moisture content under which repellency can occur, seem more reliable (Doerr et al., 2006; Hubbert et al., 2006; Scott, 2000; Zavala et al., 2009).

The objective of this study was to address the lack of knowledge on WR variations for calcareous Mediterranean forest soils and to estimate the probability of WR occurrence considering these variations. More specifically it focuses on the occurrence and persistence of WR under pine forest affected by wildfires at different times in the past and examines its spatial variability at different scales: i) the macro-scale between 4 sites at different stages of plant succession after fire; ii) the meso-scale within the same site between vegetation type and within plots of the same vegetation type, and iii) the micro-scale between 10 measurements within the same 10 cm × 10 cm plot, examining also the monthly changes with moisture content. The data collected also was used to produce exploratory models for predicting the probability of the natural background (not fire-induced) WR to occur using parameters that are straightforward to determine: vegetation type, moisture content and soil depth.

2. Material and methods

2.1. Study area and site selection

The study was conducted in the Sierra de Enguera Mountain range, eastern Spain (Fig. 1), which has a Mediterranean climate (Csa; Köppen, 1931). The mean annual temperature is 14.3 °C, with January being the coldest month (7.3 °C daily average) and August the hottest (23 °C). The mean annual precipitation is 551 mm, with a dry summer, <57 mm between June and August. The largest rainfall events are between September and December, when rainfall intensities of 100 mm day^{-1} can occur (data from the Enguera observatory (1971–2010) at approximately 15 km from the study sites). The parent material is composed of Cretaceous carbonate rocks, mainly limestone, calcarenite and marls. The soil is covered with abundant rock fragments (35%) and the soil depth is variable. The dominant soil in the study area is Xerorthent, but in some locations a deeper Alfisols (Rhodoxexalf and Haploxeralf, commonly called *terra rossa*) can be found (Soil Survey Staff, 2006). The overstorey vegetation is composed of *P. halepensis* trees either from plantation or natural regrowth after previous forest fires. Shrubs and grasses are mainly *Erica multiflora, Pistacia Lentiscus, Q. coccifera*, *R. officinalis* and *Brachypodium retusum*.

Measurements were carried out at four sites of around 1 ha each called $F_1$99, $F_2$79, $F_3$08, and $F_4$79 with the number identifying the year when the site experienced the last forest fire (see Table 1). Sites were adjacent to each other by pairs F_1/F_2 and F_3/F_4 and the distance between both pairs was 5 km (Fig. 1). All of the sites had the same vegetation and soil type and the fires had been of medium-high severity. Fire severity was estimated by the ash colour and other post-fire indicators described in Keeley (2009) from direct observations for the fire in 2008 and, from descriptions (personal communications and ADENE's (2010) reports) for the 1991 and 1979 fires. The main differences between sites were related to the recovery time since the last fire; i.e. the age and density of the vegetation, litter thickness and some slight differences in the soil properties (Table 1).

2.2. Soil water repellency and soil moisture assessment

Each site was sampled 18 times (once per month and twice in July 2008), from April 2008 to July 2009 and one final time in August 2011. For logistical reasons, not all four sites could be sampled in the same day. On those occasions, data was collected over two successive days in order to record the WR measurement data under the most similar atmospheric and soil moisture conditions possible (Doerr et al., 2002).

WR measurements were taken in the field using the WDPT (Water Drop Penetration Time) test, which involves placing a water drop of distilled water (~0.05 mL) on a soil surface and recording the time in seconds until its complete penetration (Doerr, 1998). Five classes of WR persistence were used for WDPT data classification: class 1, wettable (WDPT ≤ 5 s); class 2, slightly water repellent (5–60 s); class 3, strongly water repellent (60–600 s); class 4, severely water repellent (600–3600 s); and class 5, extremely water repellent (> 3600 s) (Bisdom et al., 1993). This categorical classification was used in the first part of this study while in the second part, where

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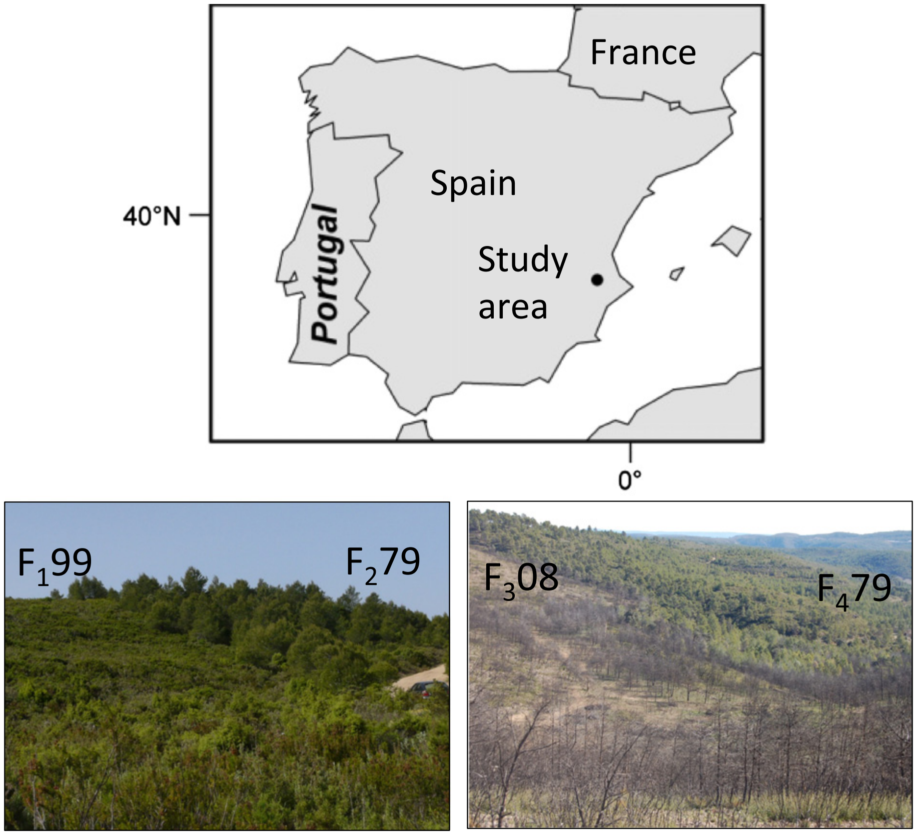

Fig. 1. Location of the study areas and sites sampled.

the probability of WR occurrence was examined, we used a dichotomic categorization, i.e. wettable (WDPT ≤ 5 s) or water repellent (WDPT > 5 s).

In each of the 4 sites, 5 different micro plots (10 cm × 10 cm) were selected for each of the three vegetation types: *P. halepensis, Q. coccifera, R. officinalis* plus 5 bare soil plots. The sampling was a simple random, avoiding any criteria at the moment of the sample selection, and carried out at different micro plots every month due to the destructive nature of the analysis. In site $F_1$99, sampling points under *P. halepensis* were not included due to the reduced number and small size of the trees regenerated after the fire in 1991. At that time, trees had still been in the juvenile stage and had not provided an adequate seed bank for regeneration (Verkaik and Espelta, 2006). In addition, sampling plots in bare soil in sites $F_2$79 and $F_4$79 neither were included due to the absence of bare soil areas. Inside each micro plot of 10 cm × 10 cm, 10 WDPT measurements were performed at the soil surface and at 1 cm depth. At each plot, two soil samples (from 0 to 1 cm for the surface sample, and from 1 to 2 cm for the 1 cm depth sample) were taken and brought to the laboratory in sealed containers to allow subsequent determination of soil water content (Fig. 2). This was determined gravimetrically by quantifying the water loss from subsamples following drying at 105 °C for 24 h.

2.3. Statistical analysis

The statistical analysis comprised two different parts. The WDPT data was not normally distributed and we did not perform common transformations to achieve normality (logarithm or inverse), due to the high number of zero WDPT values in the data set. Non-parametric approaches were therefore used.

In the first part, we conducted an exploratory analysis to examine the temporal and spatial variability of WR at different scales using the

Table 1
Study sites characteristics.

Site name	Coordinates, altitude (m)	Year of the last fires	Ha burned[a]	Average height of the tallest vegetation (m)	Average litter thickness (mm)	Soil properties[b]					
						Texture[c] (% sand, silt, clay)	SOM (%)	pH	EC (µS cm^{-1})	Munsell Colour	$CaCO_3$ (%)
$F_1$99	(38° 55′ N, 00° 50′ W) 750	1979 1999	44000 3200	1.2	2 Not continuous	(47, 28, 25)	7.6	8.5	132	10 YR 3/2	47.63
$F_2$79	(38° 55′ N, 00° 50′ W) 750	1979		12	20	(43, 30, 27)	10.1	8.5	156	10 YR 4/4	43.13
$F_3$08	(38° 55′ N, 00° 54′ W) 850	1979 April 2008	44000 89	0.5	2 Only under regenerated vegetation	(37, 39, 24)	9.9	8.3	246	7.5 YR 3/2	37.82
$F_4$79	(38° 55′ N, 00° 54′ W) 850	1979	44000	12	30	(34, 47, 18)	11.7	8.2	160	7.5 YR 3/3	26.72

[a] All the fires burned at medium-high intensity.
[b] Values based on pooled sample material from 15 sampling Points. SOM: Soil organic matter; EC: Electrical conductivity.
[c] Sand: 2–0.05 mm; silt: 0.05–0.002 mm; clay: <0.002 mm.

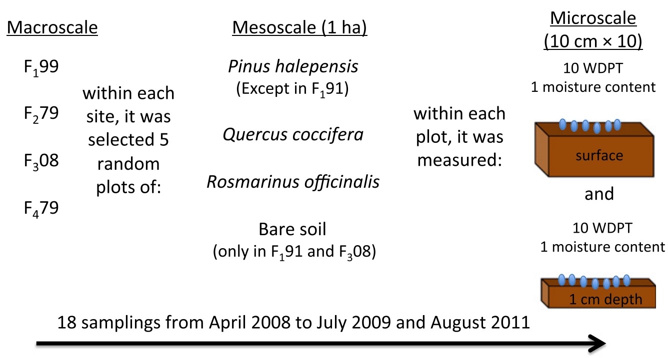

Fig. 2. Sampling strategy. For information about the sites see Table 1 and Fig. 1.

median of the 10 WDPT measurements in each plot paired with the respective moisture content for a given depth, using the 5 WR persistence classes considered. The Kolmogorov–Smirnov two-sample test (as used by Doerr et al., 1998) was chosen because it is one of the most suitable and simple nonparametric tests to assess the differences in frequency distributions data. P values for significance were lower than 0.001 unless otherwise stated.

The second analysis consisted in the construction of a model to estimate the probability of WR to occur using the dichotomic categorization of WDPT (i.e. present/absent). For that, we used individually the 10 WDPT measurements from the same plot of 10 cm × 10 cm associated with one moisture content measurement. The model was performed only in sites $F_2$79 and $F_4$79. These two sites were well suited for this analysis as both had a forest fire in the same year and were characterised by similar and already stable vegetation recovery, therefore avoiding additional effects of recent fire on WR.

Mixed-Effect Logistic Regression Models for assessing the probability of WR were considered for each site within the general framework of Generalised linear mixed models (Faraday, 2006; Hedeker, 2005). The set of explanatory variables considered was always soil moisture as covariate and vegetation type and soil depth as fixed effect factors. In addition, the experimental split-plot design (Quinn and Keough, 2002) used to collect the data introduced other elements to consider in the statistical model, specifically random effect factors: (i) the plots for each vegetation type are selected randomly and in the same plot only grows one particular vegetation type, and (ii) the measurements in surface and at 1 cm depth are related because they come from the same plot, involving that the differences between them are only due to the factor soil depth and not as a consequence of any difference between the plots.

The *Logit* function was used as the link for connecting the probability of WR to the explanatory variables. The Akaike Information Criteria (AIC) and the Bayesian Information criterion (BIC) were considered for the model selection (Akaike, 1974; Johnson and Omland, 2004). Both criteria together describe the goodness fit of the model and its complexity assuming the former quality as a good property of the model, but not the latter, according to the principle of parsimony. Deviance is also considered because it has some analogies to the residual sum of squares and the coefficient of determination in linear models (Wood, 2006). Estimation and comparison of all the models were made through an lmer function from the lme4 Package in R (De Boeck et al., 2011).

3. Results

3.1. WR variability between the study sites

More than 60% of the measurements of WDPT showed a wettable soil status for the four sites studied over the whole period considered, irrespective of the moisture content and vegetation type. However, WR was found in all the sites at some stage, both on the soil surface and at 1 cm depth, and with the exception of site $F_1$99, WR persistence was spread over all the 5 classes, including the most severe with penetration times exceeding 3600 s (Table 2).

Site $F_1$99 had the lowest WR occurrence and persistence, both at the surface and at 1 cm depth (less than 10% occurrence) and paired comparisons shown that its WR distribution was significantly different from sites $F_2$79, $F_4$79 and $F_3$08 ($P<0.001$). WR was most widespread and persistent at the surface for the two sites burned in 1979 (40% occurrence) and lower at 1 cm depth (25% occurrence), but with no statistically significant difference between them either at the surface or at 1 cm depth. These two sites did show significant differences with $F_3$08 at the surface (17% occurrence), which also showed significant differences between the surface and 1 cm depth (27% occurrence).

3.2. Temporal variability of WR within the sites studied

WR occurrence and persistence changed with moisture content; i.e. it varied with the period of the year considered. WR was absent for soil moisture values higher than 23.3% at site $F_1$99, 42.7% at $F_2$79, 56.3% at $F_3$08 and 45.1% at $F_4$79. WR seemed to follow a recognisable pattern with soil moisture (Fig. 3 provides a good example for the soil surface at site $F_4$79); however, apparently there was no distinct relationship between the occurrence, persistence and the soil moisture. A similar pattern to site $F_4$79 occurred in $F_2$79 and, with lower WR values at both sites at 1 cm depth and at site $F_1$99 for both depths (20% occurrence maximum during the summer months). In contrast, such a pattern was not evident at site $F_3$08 (Fig. 4). WR values were similar to $F_2$79 and $F_4$79 at the surface immediately after the 2008 fire, but were reduced the next summer significantly (from as much at 40% repellent samples to 10%), and were still low in August 2011. At 1 cm depth WR occurred more frequently. It was spatially continuous immediately after the fire in April 2008,

Table 2
Relative frequency distribution (%) of the WR persistence classes grouped by sites, for the soil surface and 1 cm depth. Each WDPT value represents the median of ten drops of the same plot (The number of data points for sites $F_1$99, $F_2$79 and $F_4$79 was n = 540 and for site $F_3$08 n = 720).

	Site Name	Relative frequency (%) distribution of WR persistence					Significantly different from[a]:
		Wettable	Slightly	Strongly	Severely	Extremely	
(a)	$F_1$99, surface	93.4	6.1	0.5	0.0	0.0	b, d
(b)	$F_2$79, surface	64.7	14.0	16.9	3.9	0.5	a, c, g
(c)	$F_3$08, surface	82.8	6.6	7.8	2.2	0.6	b, d
(d)	$F_4$79, surface	65.3	17.1	11.1	6.0	0.5	a, c
(e)	$F_1$99, 1 cm depth	96.8	3.2	0.0	0.0	0.0	f, g, h
(f)	$F_2$79, 1 cm depth	78.3	14.3	5.6	1.2	0.6	e
(g)	$F_3$08, 1 cm depth	71.6	15.8	10.9	1.1	0.7	e, b
(h)	$F_4$79, 1 cm depth	74.1	12.7	9.0	3.6	0.6	e

[a] Kolmogorov–Smirnov two-samples test; $P<0.001$ between samples from the same soil depth or the same site. It is not mixed site and different depth.

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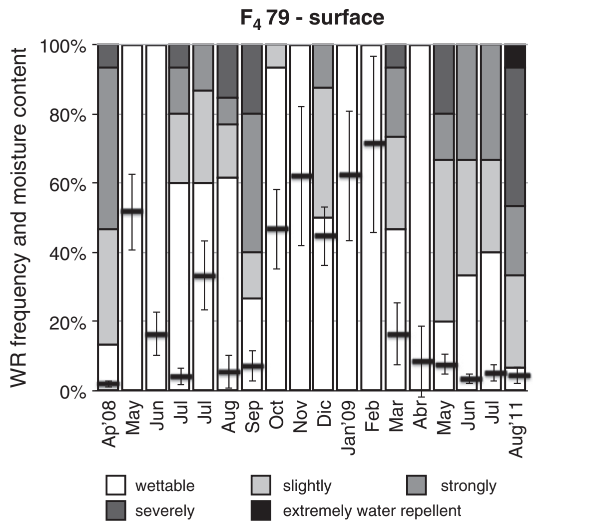

Fig. 3. Relative frequency distribution of the number of plots falling into each WR persistence class grouped by months for the soil surface at site $F_4$79 (n = 270). Each WDPT value represents the median of ten drops within the same plot. Mean values of soil moisture (horizontal bars) and corresponding standard deviations (vertical lines) are superimposed on the compound bars.

was more persistent and present at higher moisture contents than at the surface (Fig. 4). WR at depth was also reduced after the winter and, either at the surface and 1 cm depth.

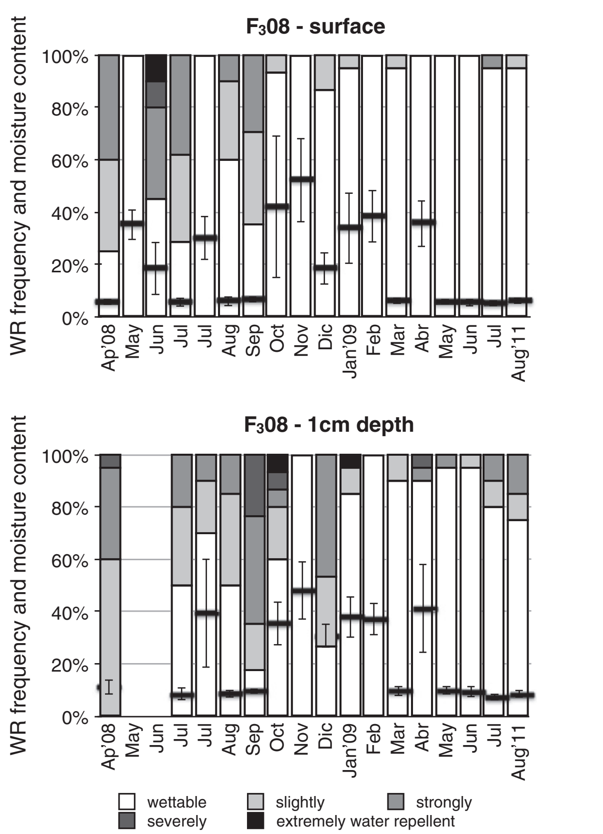

Fig. 4. Relative frequency distribution of the number of plots falling into each WR persistence class grouped by months for the soil surface (above) and 1 cm depth (below) at site $F_3$08 (n = 360 for each graph). Each WDPT value represents the median of ten drops within the same plot. Mean values of soil moisture (horizontal bars) and corresponding standard deviations (vertical lines) are superimposed on the compound bars.

3.3. Occurrence, persistence and spatial variability of WR within the sites studied

Vegetation recovery at three of the four sites was at a different stage of the post-fire succession and WR varied between vegetation types differently at each site (Fig. 5). Site $F_1$99 had the lowest WR occurrence and persistence for *R. officinalis* and *Q. coccifera* and especially for bare soil, with 5% of WR occurrence for the whole period. No statistically significant differences in WR were found between the shrubs, only between shrubs and bare soil. Regarding sites $F_2$79 and $F_4$79, despite the differences observed between each vegetation type, their WR persistence class distribution was not statistically different when compared between pairs, either at the surface, at depth or between surface and 1 cm depth for each type. At $F_3$08 site, significant differences between the WDPT distribution classes and vegetation types only were detected at the soil surface between the vegetated areas and bare soil, at 1 cm depth between *Q. coccifera* and *P. halepensis*, and between the soil surface and 1 cm depth for bare soil.

WR showed a large variability within the 5 plots of the same vegetation type sampled on the same day (Fig. 6). WDPT varied between four classes for sites $F_2$79, $F_3$08 and $F_4$79, which is the same range of variability as found between different vegetation types. The cases that only presented one WR persistence class for the 5 plots of the same vegetation type corresponded 89% of the time to the class 'wettable', with a moisture content above the threshold for each site, as noted in the previous section. Thus, variability decreased as water content increased. The lowest variability ($P<0.001$) for the same vegetation type was detected at $F_1$99, where most of the samples were also wettable even at lower moisture contents. Paired comparisons showed no statistically significant differences regarding the WR persistence variability (in terms of number of WDPT classes) within the same vegetation type among *P. halepensis, Q. coccifera* and *R. officinalis* either at the surface or at 1 cm depth. The exception, with a significantly lower variability compared to the other vegetation types, was for bare soil. Bare soil produced the largest number of wettable samples (Fig. 4). For site $F_3$08, variability within the same vegetation type was significantly reduced for the second year (2009) and WR was spread over no more than three different classes during 2009.

3.4. Micro-plot variability

Considering the WR persistence class of each of one of the 10 drops within the same 10 cm × 10 cm plot under a specific vegetation type (Fig. 2), we can observe a large variability of WR persistence classes (Fig. 7). The maximum variability occurred at the two sites burned in 1979 ($F_2$79 and $F_4$79) and the minimum in $F_1$99, where 90% of the drops inside the 10 cm × 10 cm were in the same class (wettable; Fig. 7). Statistically significant differences ($P<0.001$) for paired comparisons in the distributions of number of classes of WDPT among sites were only found between $F_1$99 and the rest of the sites and, again in $F_3$08 between the samples from years 2008 and 2009. None of the vegetation types, either at the surface or 1 cm depth seemed to present more intra plot variability than the others within the same site (e.g. site $F_2$79; Table 3), except at $F_3$08 for bare soil (always wettable) and the rest of vegetation at the soil surface, and for bare soil at the surface and at 1 cm depth.

3.5. Construction of the final selected models for the estimation of soil WR probability of occurrence

Despite the fact that the two sites considered for developing the model, $F_2$79 and $F_4$79, were burned in 1979 and the vegetation recovery was similar, the soil WR interactions with soil moisture, vegetation type and soil depth during the period studied was not similar

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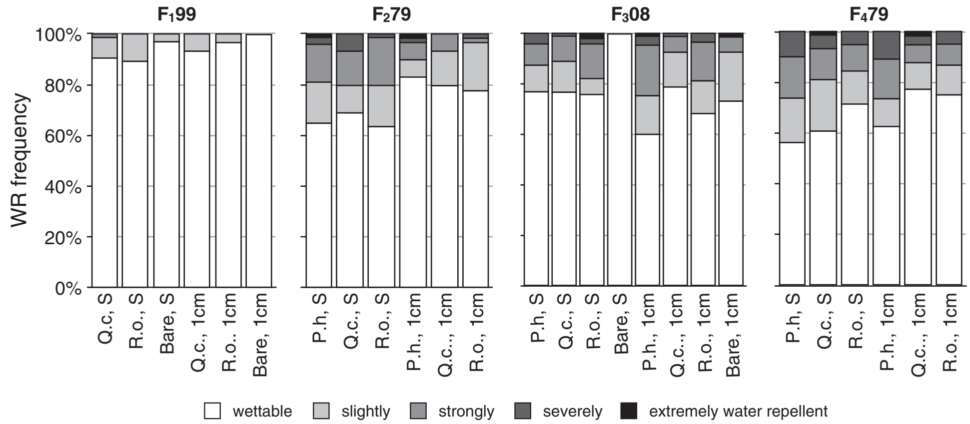

Fig. 5. Relative frequency distribution of plots falling into each WR persistence class grouped by vegetation species and soil depth at sites $F_1$99 (n = 540), $F_2$79 (n = 540), $F_3$08 (n = 720) and $F_4$79 (n = 540). Abbreviations are: *Pinus halepensis* (P.h.), *Quercus coccifera* (Q.c.), *Rosmarinus officinalis* (R.o.), bare soil (Bare) at the surface (S) and 1 cm depth (1 cm). Significant differences (P < 0.001) occurred only between P.h. S, Q.c. S, R.o. S and Bare S; Ph 1 cm and Qc 1 cm and Bare S and Bare 1 cm. Each WDPT value represents the median of ten drops within the same plot.

enough. Therefore each site was treated individually and two different models were obtained.

The statistical analysis started with the more complex model, which included the effects of the vegetation type, soil depth and soil moisture together with all the possible interactions and random effects for each plot and the interaction between the plots and measurements in surface. From this initial point, we obtained new and simpler models by comparing and selecting the least significant elements and removing or transforming them. The process continued until we obtained the simplest and most significant model with the minimum AIC (Akaike Information Criteria) and BIC (Bayesian Information criterion). These indicators describe the goodness fit of the model and its complexity between models obtained from the same data.

The final selected model for site $F_2$79 only contains as significant factors the fixed effect factor associated to soil depth (surface and 1 cm) and the covariate soil moisture, indicating that the types of vegetation did not have a significant effect on the probability of WR, as also indicated previously (in Section 3.3) from the outcomes of the Kolmogorov–Smirnov test (Table 4). Moisture and soil depth had opposite significant effects: increasing values in soil moisture produced, in general, a decrease in the probability of repellency being higher at the surface than at depth. Both factors did not have a relevant interaction term. The importance of each variable explaining the occurrence of WR is indicated by the estimated coefficient and the P value, which indicated its significance (Table 4). The estimated standard deviation associated with the plots was $\sigma_P = 5.4062$ and

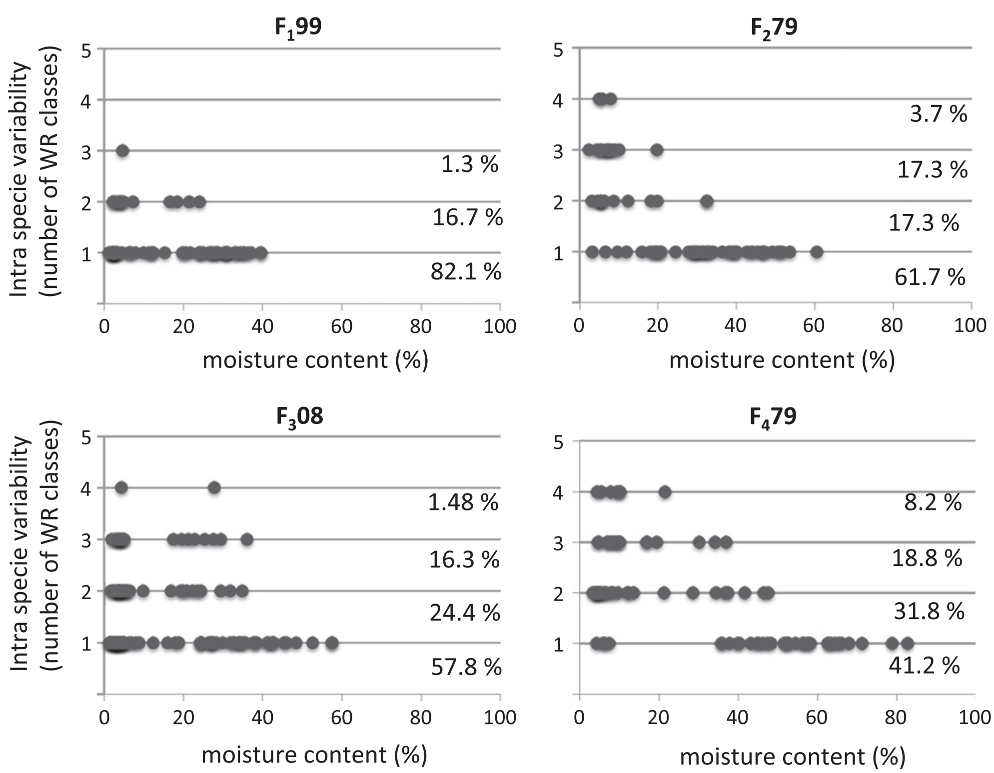

Fig. 6. Moisture content and WR variability intra-species expressed as the number of WR persistence classes within each of the 5 plots with the same vegetation type from the same site sampled on the same day. Moisture content is the average of the 5 values measured for each plot. (The number of data points at sites $F_1$99, $F_2$79 and $F_4$79 was n = 540 and for $F_3$08 n = 720).

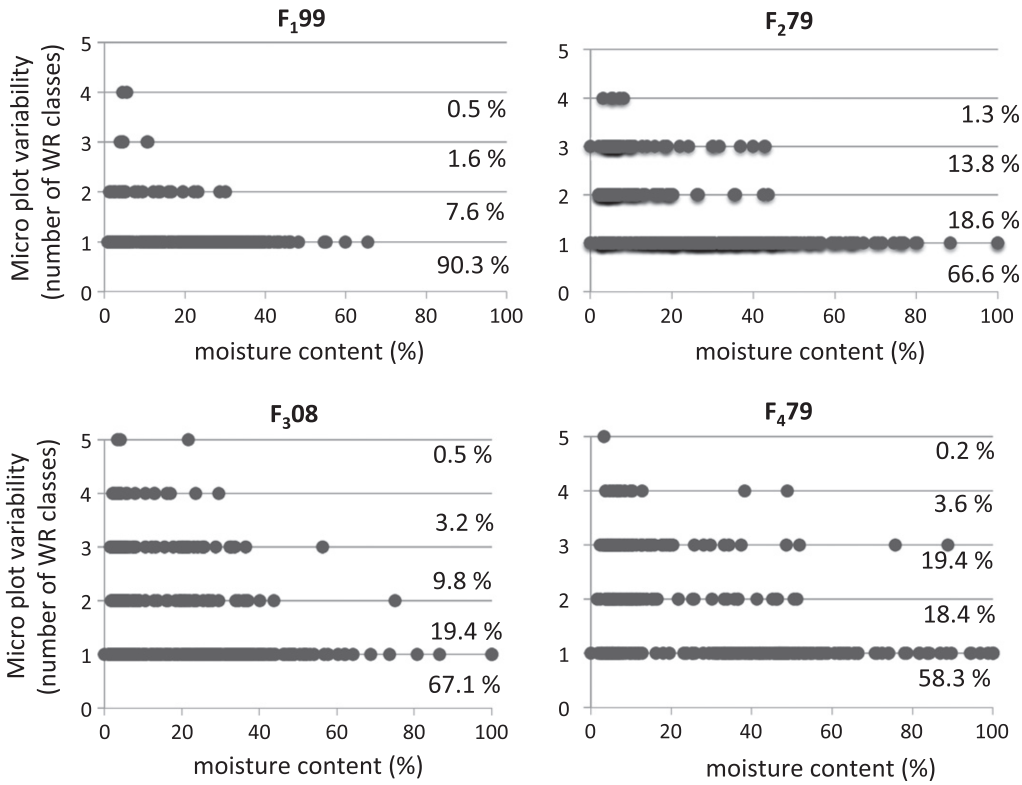

Fig. 7. Moisture content and WR variability at microscale expressed as the number of WR persistence classes within the ten WDPT values in the 10 cm × 10 cm plot areas for the different sites. There is one moisture content value per 10 WDPT measurements. (The number of readings for sites $F_1 99$, $F_2 79$ and $F_4 79$ was $n_{WR} = 5400$, $n_{moisture\ content} = 540$ and for site $F_3 08$ $n_{WR} = 7200$, $n_{moisture\ content} = 720$).

the associated variability of the random interaction between each plot and soil depth was also relevant, but somewhat smaller ($\sigma_{PxS} = 3.6148$). These results indicate a great heterogeneity of WR occurrence between the different plots and also, although to a lower degree, between the surface and 1 cm depth of each plot. AIC, BIC and Deviance values for this model are 1410, 1447 and 1398, respectively.

The final selected model for site $F_4 79$ was more complex. It included, as significant variables, soil depth and soil moisture as in site $F_2 79$, but also vegetation and some relevant interactions (Table 5). Thus, the probability of WR to be present under *P. halepensis* was higher than under the other two vegetation types, which were classified in a single group because there were no differences between them. Moisture and surface have a similar behaviour than for site $F_2 79$, reducing and increasing, respectively, the probability of WR. Moisture and vegetation also present a negative, but small, interaction. The estimated standard deviation associated with the plots was $\sigma_P = 2.2226$ and the associated variability of the random interaction between each plot and the factor soil depth was $\sigma_{PxS} = 3.7509$. These results indicate again the important heterogeneity of the probability of WR between the different plots. However, in comparison with $F_2 79$ it is about half of it or in other words that variability between each plot within site $F_4 79$ was lower than in $F_2 79$. The variability associated with the interaction between plot and soil depth was practically the same for both sites. The values here obtained for AIC, BIC and Deviance are 2266, 2323 and 2248, respectively. These values were higher than the corresponding values in site $F_2 79$ due to the more complexity of the estimated model for $F_4 79$.

Note that in the previous sections we did not consider WDPT in this dichotomic form and consequently, the results in this section will not be fully comparable to those given in Section 3.3.

Table 3
WR variability at microscale expressed as number of WR persistence classes within the ten WDPT values in the 10 cm × 10 cm plot areas for the different sites grouped by vegetation species and soil depth for site $F_2 79$ ($n_{WR} = 5400$, $n_{moisture\ content} = 540$).

$F_2 79$	Vegetation type	Number of WR persistence classes within the micro plot					Significantly different from[a]:
		1	2	3	4	5	
(a)	*Pinus halepensis*, surface	58.1	23.0	16.2	2.7	0.0	–
(a)	*Quercus coccifera*, surface	63.5	24.3	10.8	1.4	0.0	–
(c)	*Rosmarinus officinalis*, surface	60.8	25.7	13.5	0.0	0.0	–
(d)	*Pinus halepensis*, 1 cm depth	77.6	10.3	8.6	3.4	0.0	–
(e)	*Quercus coccifera*, 1 cm depth	67.8	13.6	18.6	0.0	0.0	–
(f)	*Rosmarinus officinalis*. 1 cm depth	74.1	10.3	15.5	0.0	0.0	–

[a] Kolmogorov–Smirnov two-samples test; P<0.05 between samples from the same soil depth or the same site. It is not mixed site and different depth.

3.6. Estimation of the occurrence probability of soil WR for a long-unburned Mediterranean calcareous forest

The estimated probability of finding WR at site $F_2 79$ (surface and 1 cm depth) for the selected Mixed-Effect Logistic Regression Model showed a decrease with the percentage of soil moisture. The

Table 4
Estimated results associated with covariate soil moisture and fixed-effect elements in the selected Mixed-Effect Logistic Regression Model for the probability of water repellency at site $F_2 79$. Level depth is the reference class named intercept in the table.

	Estimate	Std. error	*Z* value	P value
Intercept	−1.6051	1.1790	−1.361	0.1734
Moisture	−0.3188	0.0856	−3.726	0.0002
Surface	5.5592	1.1938	4.657	0.0000
Moisture × surface	−0.03261	0.0830	−0.393	0.6944

Table 5
Estimated results associated with covariate soil moisture and fixed-effect elements in the selected Mixed-Effect Logistic Regression Model for the probability of water repellency in site $F_4$79. The combination of the categories *Quercus coccifera* and *Rosmarinus officinalis* together with depth is the reference class named intercept in the table.

	Estimate	Std. error	Z value	P value
Intercept	−0.2934	0.6212	−0.472	0.6366
Moisture	−0.16827	0.0320	−5.255	0.0000
Pinus	2.0144	0.3753	5.368	0.0000
Surface	1.9972	0.7376	2.708	0.0068
Moisture × pinus	−0.04608	0.0170	−2.711	0.0067
Moisture × surface	0.02163	0.0344	0.629	0.5292
Pinus × surface	−0.9572	0.4730	−2.024	0.0430

probability becomes practically zero at both depths for moisture values greater than 30% (Fig. 8). For moisture levels lower than 20%, a large difference in the estimated probability of WR between surface and depth is apparent. It is very close to 1 at the surface when the soil is very dry and shows a great dependence on soil moisture. On the contrary, the estimated probability of WR at depth is always small.

The general behaviour of the estimated probability of WR at site $F_4$79 in relation to soil moisture was similar to site $F_2$79 for each one of the four groups considered (Fig. 8), but in this model, the soil moisture threshold to approach the estimated probability of zero is higher. Also here there was an important and positive difference between the estimated probability of WR at the surface and at depth for both types of vegetation, but lower differences can be observed within the *P. halepensis* plots. *P. halepensis* had higher values of the estimated probability than plots of *Q. coccifera* and *R. officinalis*. However, site $F_2$79 seemed to present a more extreme behaviour for the estimated probability of WR, having most of the lower values at depth, but also the most of the higher values at the surface.

4. Discussion

4.1. Soil water repellency at sites with different burn history

The WR determined in the present study for calcareous Mediterranean forest soils under dry conditions was not as continuous and persistent as those reported from acidic and soils in heathlands in Spain, *Eucalyptus* or mixed forest in Portugal, Australia, North-America and Northern Europe (Buczko et al., 2005; Dekker et al., 2001; Doerr et al., 1998, 2009b; Stoof et al., 2011), but it was similar to other obtained under *P. halepensis* in the same soils (Arcenegui et al., 2008; Mataix-Solera et al., 2007).

Immediately after the fire, persistence of surface WR at site $F_4$08 was similar to that of the long unburned sites, but greater at 1 cm depth, presumably increased by the translocation of hydrophobic substances during burning (DeBano et al., 1970). One year later, however, WR was lower than at the long unburned sites, circumstance as also reported by Tessler et al. (2008) for a calcareous Mediterranean site 33 weeks after burning. Our observations during the fieldwork point to the same mechanisms Hubbert and Oriol (2005) suggested to be responsible for WR reduction a year after fire: the deposition of eroded soil and sediments (in our case due to water erosion and not dry ravel as found by Hubbert and Oriol (2005)) that covered the original soil surface in some places to a depth of 5 cm, combined with the lack of new litter and plant cover to provide a new influx of hydrophobic compounds. The sediment was composed mainly of wettable ash that can reduce WR when mixed with the soil. Specifically only 12% of the ash sampled in $F_3$08 was found to be water repellent (Bodí et al., 2011).

Three years later (August 2011), WR at $F_3$08 was still below that of the long unburned forest sites and it is unclear when WR would be re-established to normal 'background' levels. For the site $F_1$99, which was examined 10 years after the last fire, WR was still below the values observed at the long-unburned sites, which matches findings reported by Cerdà and Doerr (2005). *P. halepensis* was not abundant

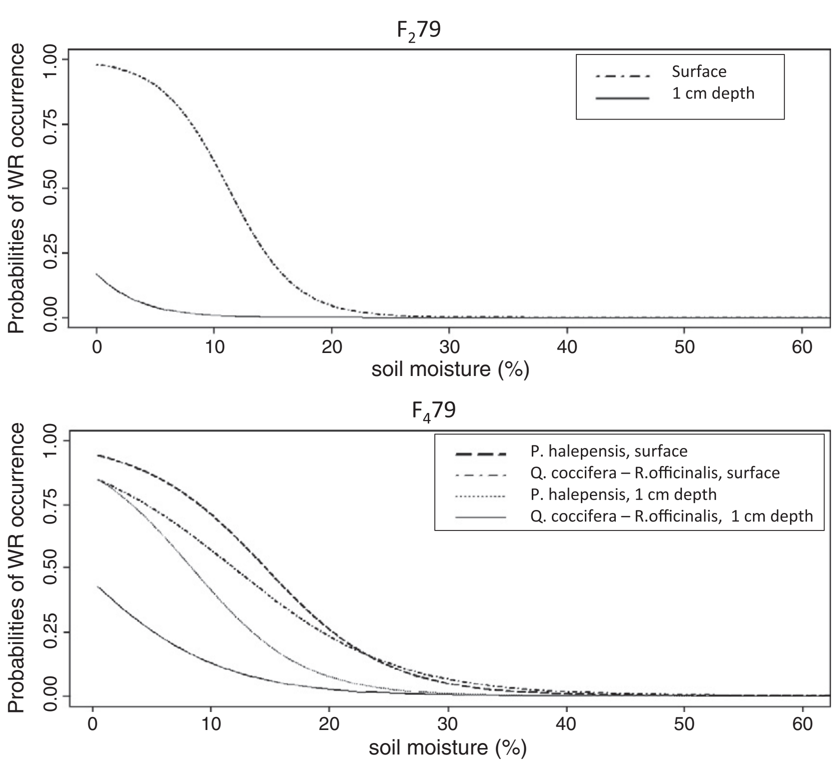

Fig. 8. Estimated probability of finding WR at site $F_2$79 (above) and $F_4$79 (below) with regard to the values of soil moisture for each one of the significant groups in the corresponding selected Mixed-Effect Logistic Regression Model.

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and still very young at this stage, however, considering that *Q. coccifera* and *R. officinalis,* which was present here, was also associated with substantial WR at other sites, the lack of mature *P. halepensis* is unlikely to be the reason. It might be that the low biomass production, leading to a patchy litter layer of only 2 mm (Table 1), did not supply a sufficient input of hydrophobic substances. In addition, the calcareous soils did not promote fungal growth as in acidic soil, where Jordán et al. (2010) found in a Spanish heathland a complete recovery of WR in 18 months despite an appreciable organic layer was not generated.

If we consider each site as representing the same type of soil ecosystem at different stages following fire, a pattern of increased WR after a fire, followed by rapid decay and a longer-term natural reestablishment emerges, which fits with the conceptual model suggested by Malkinson and Wittenberg (2011). Unfortunately there have been no observations of burned sites that were long enough to confirm this pattern. However, it will not be universally applicable as the fire may also eliminate any pre-fire WR if soil temperatures exceed the threshold for WR destruction (Doerr et al., 2009a).

4.2. Spatial variability of soil water repellency within the same site

Differences in WR persistence in relation to vegetation type for the whole period considered were not significant within our study sites (Fig. 6). Only in terms of WR occurrence were found significant differences between *P. halepensis* and *R. officinalis*–*Q. coccifera* in $F_4$79, one of the two long unburned sites studied for modelling the probability of WR occurrence (Fig. 8). The reasons of the small differences in WR persistence below different vegetation type are the high variability of WR within the same species. The variability between vegetation types is thus not higher than the variability within the same vegetation type in any of the sites. High variability in WR persistence under the same vegetation was also found in Portugal by Stoof et al. (2011) within the same transect in a *Eucalyptus globulus* and by Doerr et al. (1998) in burned *Pinus pinaster* stands. Only significant differences were found between the vegetated and bare soils. In other studies, WR has also been reported to be higher under the canopy than at the canopy edge for burned and unburned soils (Cerdà et al., 1998; Gimeno-García et al., 2011; Keizer et al., 2005; Madsen et al., 2011). Exceptionally WR was registered for bare soil at site $F_3$08 at 1 cm depth and was probably induced by fire.

In the long unburned sites there was a continuum of vegetation cover and litter layer derived from a variety of species homogenising the possible different effects of each vegetation type on WR. Field observations also indicated that if fungal hyphae existed, WR was present independently of the vegetation type as also found in previous studies (Arcenegui et al., 2008; Crockford et al., 1991; Gimeno-García et al., 2011; Hubbert et al., 2006; Scott and van Wyk, 1990). In addition, the factors that are likely to affect WR within the same site, including litter depth, soil microorganisms and soil properties, are not always spatially homogeneous (Bochet et al., 1999; Cerdà, 1997; Doerr et al., 2000; Spielvogel et al., 2009). Moisture content in particular, can be very inhomogeneous and change even daily (Rodriguez-Iturbe et al., 1995). This discontinuity means that WR is unlikely to be a spatially continuous property within the same site. Furthermore, for the recent fire affected site, the spatial variability in soil heating during burning of different patches of vegetation and litter, may lead to enhanced or destroy WR (Gimeno-García et al., 2011; Scott and van Wyk, 1990), causing further variability and, also the ash deposited during burning can vary in wettability, with subsequent effects on soil when incorporated in the soil (Bodí et al., 2011). The lowest spatial variability of WR reported from forests in the literature was under unburned *E. globulus* plantations in Portugal (Doerr et al., 1998). The extreme and homogenous WR was attributed to the uniform tree age, density and planting pattern together with the high biomass productivity associated with the wet Mediterranean climate.

4.3. Spatial variability of soil water repellency at the micro-scale

Few studies have focused on the variability of WR at the micro-scale (cm–mm). However, it can give insights into the reasons for the variability at larger scales and, according to Hallett et al. (2004), it can affect the sorptivity at centimetre scale. Hubbert et al. (2006) found that the total variation within 20 drops in a given area of 15 cm × 15 was as large as the variability seen over the 1.28 ha watershed. The total variation caused by within site variability seemed to decrease with soil depths. In addition, very little evidence of spatial autocorrelation was found at this scale (Hallett et al., 2004). In our study, the variability within the 10 drops in the 10 cm × 10 cm spanned 4 classes for low soil moisture contents.

Reasons for the variability at the micro-scale could be submillimetre spatial variability of organic matter and of the microbial environment in soil (Nunan et al., 2002). Since fungal biomass is spatially patchy (Ritz, 2007) it is reasonable to assume that the deposition of hydrophobic compounds that induce WR will be patchy, both at the soil surface and within the soil profile, an even inside the structure of aggregates (Urbanek et al., 2007) or within micronscale regions of and individual soil particle as indicated by Cheng et al. (2009) using Atomic Force Microscopy. According to Ritz (2007), releasing fungal hydrophobins is a strategy of fungi to compete with other groups of microorganisms, reducing the rate at which water enters soils controlling the distribution of water and inducing spatiotemporal heterogeneities. Indeed we observed in the field that where hyphae were present under any of the species, WR was detected, but varied even from wettable to extreme water repellent in a few millimetres distance.

4.4. Estimation of the probability of soil WR occurrence for a long-unburned Mediterranean calcareous forest

Few statistical models attempting to examine the behaviour of WR in connection to relevant covariates and factors have been derived to date. This is not surprising due to the WR variability already discussed and the complex set of potential explanatory variables for WR reported in the literature. Relationships of WR with organic matter, soil texture or moisture content supposedly affecting WR were found to be not consistent (see review by Doerr et al., 2000) and Doerr et al. (2006) concluded, in a study of WR under a large variety of different soil and land uses types, that land use (i.e. presence and type of vegetation) and the critical water content above which WR is absent, were the most promising factor for predicting WR occurrence.

Some of the models performed in previous work analysed WR as a continuous variable with a normal distribution. Harper and Gilkes (1994) considered logarithm transformations of WR as a random normal distribution and performed a multivariate regression analysis for a range of land types in New Zealand founding that the variables amorphous iron, clay and organic matter, explained 63% of the variation in WR. They found multivariate relationships to be more powerful in explaining WR variations than bivariate regressions usually performed due to the multivariate dependency of WR. Scott (2000) considered a regression model similar to the previous for forests in South Africa using organic carbon content, specific surface area, sand and clay content as possible predictors of WR as determined by the *Apparent Contact Angle* and *Critical Surface Tension* methods. For the former method, only carbon content was a significant predictor explaining 24% of the variance and, for the latter method, organic carbon and specific surface area were both significant predictor terms in the model, which explained 46% of the variation. The study concluded that the different vegetation types explained the main

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variation in WR. Outside the analysis considering WR as a normal variable, Hubbert et al. (2006) developed a stochastic model using soil bulk density, sand and clay content, litter depth, moisture, and species type as a potential predictors and found that finding only soil moisture and species type to had a significant effect on the persistence of soil WR for unburned surface sites.

Our study is different to those carried out to date because our regression model focused on the probability of WR occurrence and not on the raw WR values and, as also done by Hubbert et al. (2006), considered non-parametric tools for the analysis. The model constructed here employed factors and covariates indicated by previous authors to be good predictors as well as being easy and economic to obtain, such as soil depth, vegetation type and soil moisture. Furthermore, the model also accounted for variability between each plot and the interactions between factors and covariates, and between surface and soil depth within the same plot.

The final models obtained for each site were the best, simplest and most significant possible with the minimum AIC and BIC values obtained of the set of all models considered after all the iterations with the factors and covariates used. One of the most powerful explanatory variables for the probability of WR to occur was the covariate moisture content ($P=0.0002$ for F_279 and $P=0.0000$ for F_479), which was inversely related to WR. From the fixed factors, the best predictors were not the same for both sites. The category soil surface (associated to the fixed factor soil depth) was powerful for F_279 ($P=0.0000$) and *P. halepensis* (associated to vegetation type) for F_479 ($P=0.0000$; Tables 4 and 5). Due to the fact that for site F_279, there were no significant differences between vegetation type and this fixed factor was suppressed during the process of obtaining the best and simplest model, the fixed factor surface was the most explicative for WR occurrence. This is to be expected since WR has been mostly reported to be more common in surface soil than at depth in many environments (Doerr et al., 2000). For site F_479, the fixed factor vegetation type, i.e. specifically the category *P. halepensis,* was more significant than the category soil surface itself. *P. halepensis* has been found in other studies focusing on this ecosystem as a vegetation type associated with particularly high WR occurrence and persistence (Cerdà and Doerr, 2005; Mataix-Solera et al., 2007). The reasons for some difference between vegetation type to occur for this site it is unclear as both sites had similar characteristics. It was related with the lower estimated standard deviations associated with plots, which indicate a lower variability between plots than for site F_279. The only observed difference between sites was in general a thicker litter under *P. halepensis* for site F_479. It is suggested here that the predictive power might be improved further by including other simple variables not quantified here such as the litter layer thickness (Arcenegui et al., 2007; Crockford et al., 1991; Gimeno-García et al., 2011). Soil moisture content was a powerful predictor for both sites, yet its effects were not exactly the same between both sites. Soil moisture alone has been found not to be a very precise predictor for WR in, for example, burnt eucalypt stands in wet Mediterranean north-central Portugal (Keizer et al., 2008), however, work under a range of vegetation types in humid temperate conditions has shown that a soil-type specific upper threshold exists above which WR is absent (Dekker et al., 2001; Doerr et al., 2006). In our model presented here, the predictive power of soil moisture was significant together with the other fixed factors for estimating general WR occurrence, but not its specific persistence.

The model derived for each site is powerful in the context applied here, and the approach described for its development could be applied elsewhere to estimate seasonal probability of WR occurrence for sites that have reached a reasonably stable stage in vegetation succession and homogeneity. Using the data from sites different to those examined here would allow validation of the model with independent data. We also suggest that the model will be improved with the use of Bayesian statistical methods, which address better the difficulties of dealing with random effects in non-normal distributed scenarios than the frequentist inference used here. Furthermore, in order to be able to estimate probability of WR of occurrence or even persistence at more temporally dynamic sites (e.g. ploughed, logged or fire-affected), it would be necessary to develop a dynamic model that includes vegetation recovery rates or microorganisms behaviour as a covariates, together with more correlated soil or vegetation properties and a better understanding of the mechanisms determining the change in WR associated with, for example, fire, vegetation recovery and microbial activity.

The exploratory work carried out here in developing a model for predicting WR occurrence shows promise. Future work involving further model development and validation seems warranted and may, for example, allow development of a software product capable of predicting WR from a relevant set of easy to determine explanatory variables. Such a product would be a useful tool for land managers in environments where the implications of WR are substantial.

5. Conclusions

In this study, we examined the spatial and temporal variability of soil water repellency (WR) for calcareous Mediterranean forest soils using four otherwise comparable sites (1 ha) that had been affected by fire in the recent past (two sites burned 2008 and 1999 respectively), or were long-unburned (two sites burned last in 1979). At the long-unburned sites, WR was found to be common at the surface (40–95% of samples), with some samples showing extreme WR persistence (WDPT > 3600 s) and generally lower WR values at 1 cm depth. At the site sampled immediately after burning (2008) WR was increased at 1 cm depth, similar to the long-unburned sites at the surface, however, at both depths WR decreased to 10% of samples for the next summer. The time necessary for WR to be re-established to pre-fire (i.e. long-unburned) conditions is not known, however, the comparable site burned in 1999, exhibited WR in no more than 20% of samples 10 years after the fire. This slow recovery of WR compared to other studies is attributed to the insufficient recovery of the original vegetation (*P. halepensis*). On a monthly basis, WR followed a general pattern of increase and decrease inverse with soil moisture for all sites.

Variability between WR under *P. halepensis, Q. coccifera* and *R. officinalis* within the same 1 ha site was as high as within the same vegetation type and as within a plot of 10 cm × 10 cm, resulting in limited differences in WR between the different vegetation types. The least variability was found in bare soil plots, which were mostly wettable.

Using covariates that are straightforward and economic to measure (vegetation type, soil depth and moisture content) we derived a statistical model that allows estimation of the probability WR occurrence (presence or absence). This is a statistically significant and simple model that may represent a powerful tool estimate WR occurrence under different seasonal and meteorological conditions in successionally mature (i.e. long-unburned) areas. Further work is required to determine the wider applicability of this model.

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6.3. The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content. Geoderma, 160, 599-607.

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The wettability of ash from burned vegetation and its relationship to Mediterranean plant species type, burn severity and total organic carbon content

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ABSTRACT

Immediately following a wildfire, a layer of ash and charred material typically covers the ground. This layer will gradually be reduced, being redistributed by rainfall, wind or animals, by partial dissolution and by incorporation into the soil. Ash can increase or decrease the post-fire runoff and erosion response, depending upon the soil and ash properties and the ash thickness. One aspect of ash that has remained unknown and which may affect the variability in the hydrological response of the burned soil is its wettability. This study examines the wettability of ash using the Water Drop Penetration Time (WDPT) test, its relationship with total organic carbon (TOC) content and colour, and its effects when incorporated into the soil. Ash samples ($n = 48$) were taken from five wildfires in the Mediterranean basin encompassing a range of burn severities. Additional ash material was generated from exposing plant leaves (*Rosmarinus officinalis*, *Pinus halepensis* and *Quercus coccifera*) to specific temperatures in a muffle furnace (200–700 °C). Selected laboratory ash types were also mixed with wettable and water repellent soil material to assess their effect on soil wettability.

Over 30% of ash samples from the wildfire sites exhibited water repellency (WR), with WDPT levels ranging from low to extreme. This variability appears to be related to differences in fire severity, combustion completeness of the biomass, vegetation type and subsequent rainfall events. The laboratory-generated ash exhibited a greater frequency and persistence of water repellency at lower furnace temperatures (200–300 °C), with ash from *R. officinalis* being generally less water repellent than that derived from the other two species. The water repellency levels of ash correlated well with TOC ($r = 0.80$), whereas neither of these parameters correlated very well with ash colour ($r = 0.57$ for TOC and $r = 0.59$ for WR). This suggests that ash colour, which is widely used as a parameter in classifying burn severity in the field, may not necessarily be a very accurate indicator. Adding water repellent ash to wettable soil induced WR, whereas the addition of wettable ash to water repellent soil had the opposite effect. A wetting–drying cycle can reduce the water repellency of a soil–ash mixture. There were substantial differences between wildfire- and laboratory-generated ash in terms of organic carbon content and colour, suggesting that the combustion conditions in a furnace may not adequately represent those in wildfires.

In contrast to what is generally assumed, our findings demonstrate that ash from vegetation fires can be water repellent. This is likely to have implications for runoff responses and nutrient fluxes not only when ash is present on the ground surface, but also following its redistribution and incorporation into the soil.

1. Introduction

After the passage of wildfire, a layer of ash and charred material often covers the ground as a result of biomass combustion processes. There is currently no widely accepted definition of wildfire ash, and in this study, ash is considered to be the solid residue from biomass burning consisting of charred organic material, charcoal and residual mineral material (Scott, 2010). The amount of ash deposited, its spatial distribution and its physical and chemical characteristics will vary depending on the type, weight, moisture and spatial distribution of the pre-fire biomass as well as the intensity, duration and micro-meteorological conditions of the fire (Raison, 1979; Ulery et al., 1993). Ash thicknesses typically range from less than 1 to 10 cm (Cerdà and Doerr, 2008; Goforth et al., 2005; Woods and Balfour, 2008), although ash thicknesses of up to 20 cm have been observed in areas of concentrated fuel combustion (Gabet and Sternberg, 2008). Ash tends

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to be distributed in patches of black, grey and white colour, although black ash is usually located also underneath a white ash layer (Blank and Zamudio, 1998; Lewis, 1974). Black ash contains a higher proportion of charred organic material than the whiter mineral-rich ash because of a less complete combustion of the former, thought to occur at lower fire intensities (Khanna et al., 1994; Knicker, 2007; Neary et al., 1999). On that basis, ash colour has been used as an indicator of fire severity (Keeley, 2009; Lentile et al., 2006; Robichaud et al., 2007). Despite this widely accepted relationship, few studies have attempted to critically examine the actual relationship between ash organic carbon content, colour and fire intensity, and those conducted have focussed on laboratory vegetation burn experiments under different temperatures (Etiegni and Campbell, 1991; Liodakis et al., 2005; Misra et al., 1993; Úbeda et al., 2009; White et al., 1972). The presence of an ash layer on a burned site is often short-lived with ash being removed off site by rainfall or wind with its soluble components being dissolved during rainfall, or incorporated into the soil by physical and bioturbation processes. However, the presence of ash on the soil surface in the initial period after a fire can be very important. Firstly, ash is a significant source of nutrients for the soil (Badia and Martí, 2003; Demeyer et al., 2001; Khanna et al., 1994; Soto and Díaz-Fierros, 1993) and of solutes in runoff (Pereira et al., 2009; Spencer and Hauer, 1991; Spencer et al., 2003). Secondly, ash covering the ground results in changes in the hydrological behaviour of the land surface and the underlying soil and can hence modify soil erosion processes (Cerdà and Doerr, 2008; Onda et al., 2008; Woods and Balfour, 2008). While the effect of ash on the nutrient budget of the landscape has received considerable attention (see review by Raison et al., 2009), its effects on soil hydrology and erosion have seen little attention until recently and reported views of its effects are conflicting. On the one hand, ash is suggested to contribute to the blocking of pores or forming a crust at the soil surface, resulting in increased surface runoff (Gabet and Sternberg, 2008; Mallik et al., 1984; Onda et al., 2008). On the other hand, its presence as a water absorbent surface layer is thought to limit or delay surface runoff (Cerdà, 1998; Cerdà and Doerr, 2008; Kinner and Moody, 2007, 2010; Larsen et al., 2009; Leighton-Boyce et al., 2007; Woods and Balfour, 2008). In addition, ash itself is a source of sediment and has been reported to promote debris flows (Bookter, 2006; Burns, 2007; Cannon et al., 2001; Reneau, 2007). However, ash also protects soil against raindrop impact and therefore can reduce rainsplash related sediment yields (Cerdà and Doerr, 2008; Kinner and Moody, 2007; Larsen et al., 2009; Woods and Balfour, 2008; Zavala et al., 2009), and its leachates have been reported to reduce soil erodibility by promoting flocculation of the dispersed clays (Giovannini, 1994; Holcomb and Durgin, 1979). This variability is suggested to be caused by (i) the variable physical and mineralogical nature of ash produced at different combustion conditions and after interacting with the atmosphere or water, (ii) the thickness of the ash layer (Woods and Balfour, 2010) and (iii) by the different geology of the sites (Balfour and Woods, 2006; Kinner and Moody, 2007). However, one aspect that has not been explored in detail to date, which could also result in different responses in runoff processes, is the wettability of ash. In most of the literature, ash is referred to as a highly wettable material (e.g. Cerdà and Doerr, 2008; Doerr et al., 2006; Etiegni and Campbell, 1991), whereas in some studies ash has also been suggested to be water repellent, albeit with no further quantification (Gabet and Sternberg, 2008; Khanna et al., 1996; Stark, 1977).

Charred residue produced by fire can contain aromatic compounds that have a lower free energy than water and therefore behave as hydrophobic materials with reduced solubility (Almendros et al., 1992; Knicker, 2007). The organic component of ash would be expected to contain these components and it is therefore reasonable to hypothesise that ash can be water repellent under certain circumstances depending on the type and amount of organic substances present. A layer of water repellent ash present on top of the soil, or incorporated into it, may therefore have a similar effect to soil being water repellent *per se* and hence enhance rainsplash erosion and overland flow, decrease infiltration and reduce water and nutrient availability for plants (Doerr et al., 2000; Robinson, 1999; Terry and Shakesby, 1993). The objectives of this study were to examine and quantify the wettability of ash and its effects on soil hydrology. Water repellency, total organic carbon (TOC) and colour analysis were carried on (i) ash collected from a range of wildfires and (ii) ash generated in the laboratory from different vegetation types and burn conditions, in order to determine the potential relationships of TOC, colour and burning conditions with the wettability of ash. We also mixed ash and soil material of differing wettability in order to explore the effect of mixing on the wettability of the resulting materials.

2. Materials and methods

2.1. Study sites and ash, vegetation and soil sampling

Ash samples were collected from five wildfires within the Mediterranean basin (Table 1, Fig. 1). Prior to the fires, all sites were dominated by *Pinus halepensis* and the understorey comprised mainly of *Quercus coccifera* (*Quercus calliprinos* in the case of site H), *Rosmarinus officinalis* and *Brachypodium retosum.* Depending on the quantity of precipitation and the time since the previous fire at the site, other species present included *Cistus albidus, Erica multiflora, Juniperus oxycedrus, Pistacia lentiscus,* and *Ulex parviflorus*. Fire severity was estimated by the colour of the ash and other post-fire indicators described in Keeley (2009). Samples were collected in the period between 27th April to 1st September 2008, and for four of the five sites within three weeks following the fire and before the occurrence of any rain. At site N (Navalón), samples were taken six weeks after the fire, a period in which there were two rain events; however, no significant ash redistribution had occurred during this time. At each site, ash was taken from underneath pine trees and understorey vegetation ($n = 48$). At site H (Haifa), fire severity was more variable than at other sites, and here sampling was not carried out randomly, but the opportunity was used to take samples from areas of low, medium and high burn severity. Following transport to the laboratory, samples were air-dried. Vegetation and soil samples for the laboratory experiments were collected from the El Teularet Soil Erosion Experimental Station (TESEES) (38° 55′ N, 00° 50′ W) at 750 m a.s.l. in the Province of Valencia in eastern Spain. The climate here is Mediterranean, with a mean annual rainfall ranging from 479 mm at Enguera-Las Arenas meteorological station to 547 mm at the Enguera-La Matea. The autumn is generally wet and the summer dry, during which wildfires are common. Average annual temperatures range from 12.7 to 14.2 °C at La Matea and Las Arenas meteorological stations (Bodí and Cerdà, 2008). The parent material consists of Cretaceous limestone and calcarenite and the soil is a Typic Xerorthent (Soil Survey Staff, 2006) with a thin litter layer (~10–30 mm thick) (Table 2). Soil samples were taken from 0 to 2.5 cm depth of the A-horizon. Live and dead leaves for generating ash in the laboratory were collected from three common plant types: *P. halepensis* (Aleppo pine), *R. officinalis* (Rosemary) and *Q. coccifera* (Kermes oak).

2.2. Vegetation and soil preparation in the laboratory

The vegetation collected was dried at room temperature and ground in a mill until it was fine and homogeneous (maximum < 1 mm of diameter) in order to maximise sample homogeneity for subsequent laboratory burning. Sample batches of approximately 10 g leaf material each were placed in crucibles (8.5 cm diameter, 3.5 cm depth) and heated in a preheated muffle furnace (Nabertherm, P320, Bremen, Germany) at 200, 250, 300, 350, 400, 450, 500, 550 or 700 °C for 20 min. Although flaming and smouldering combustion times can vary widely in the field, the chosen heating duration is within the range of combustion times reported for Mediterranean pine litter by Ormeño et al. (2009).

Table 1
Characteristics of the study sites, fires examined and number of samples taken.

Site	Code	Number of samples	Coordinates and altitude (m a.s.l.)	Geological substrate	Soil type (Soil Survey Staff, 2006)	Annual rainfall (mm); mean temperature (°C)	Overall fire severity
Albaida (Sierra del Benicadell) Spain	A	10	38° 49′ N, 0° 30′ W 450	Cretacious limestones	Lithic Xerorthent	723; 16.2	Moderate–severe
Haifa (Isfiya-Park HaCarmel) Israel	H	3	32° 44′ N, 35° 02′ E 450	Cretacious dolomites and limestomes	Typic Xerorthent	700; 17.5	Low–moderate
Llíber (Sierra del Castell de la Solana) Spain	L	10	38° 44′ N, 0° 01 W 250	Cretacious limestones	Lithic Rhodoxeralf	845; 15.8	Moderate
Navalón Spain	N	15	38° 55′ N, 00° 54′ W 850	Cretacious calcarenites, limestones and marls	Typic Xerortent	537; 12.7	Severe
Pinoso (Sierra del Reclot) Spain	P	10	38° 23′ N, 0° 57′ W 700	Jurassic limestones	Typic Xerorthent	277; 15.8	Moderate–severe

Soil samples were air-dried in the laboratory at room temperature (~25 °C) to constant weight and passed through a 2 mm sieve to obtain a homogeneous soil sample material. This soil material was initially wettable. In order to also obtain comparable water repellent soil, 50 g were heated at 250 °C for 20 min in the same crucibles types used for vegetation burning and the soils were turned over every 5 min to ensure homogeneous heating (DeBano, 1981). This heating increased the Water Drop Penetration Time (WDPT, described in the subsequent discussions) of soil samples from 2 s to 4636 s.

2.3. Sample treatment and measurements

2.3.1. Experiment 1: ash characteristics

Wettability, total organic carbon and colour of the different ash materials from the five wildfires, and of the ash created under laboratory conditions, were analysed as described below.

In order to measure water repellency (WR), all ash samples were placed in separate 50 mm diameter plastic dishes and exposed to a controlled laboratory atmosphere (20 °C, ~50% relative humidity) for one week to minimise potential effects of any variations in preceding atmosphere humidity on WR (Doerr et al., 2002). The Water Drop Penetration Time test (WDPT) (Wessel, 1988) was used to measure the persistence of WR in the ash. This involved placing three drops of distilled water (~0.05 mL) on the surface of each sample and recording the time required for their complete penetration. The average value of three drops is reported here as the WDPT value of a sample. Penetration times were classified following specific intervals, with WDPT ≤ 5 s representing wettable and WDPT > 5 s water repellent conditions (see Table 3 for details) (Doerr, 1998).

Total carbon (TC) was measured in triplicate using a "PrimacsSC Carbon Analyser" which has a precision of ±2% of the TC values obtained. Inorganic carbon (TIC) or carbonate content was obtained based on a method developed for soils and described in MAPA (1994). This method involved adding HCl (1 N) in excess to react with carbonates and assessing the remaining HCl by titration using NaOH (0.5 N) and a pH colour indicator. The precision of the method is ±0.1% and this was also carried out in triplicate. Total organic carbon (TOC) was then determined by subtracting mean inorganic carbon from mean total carbon.

Ash colour was determined using Munsell Colour Charts (Munsell Colour co., 1998). All measurements were done by the same person and under the same light conditions. Ash samples were classified as black, grey or white according to the following criteria: i) black when ash colours were G1 2/N, G1 2.5/N, G1 2.75/N or 2.5YR 2.5/1; ii) grey ash when the colours measured were: G1 3/N, G1 3.25/N, G1 3.5/N, G1 4/N, 2.5YR 3/1 or 2.5Y 4/1; iii) and white ash if colours were: G1 4.25/N, G1 4.5/N, G1 4.75/N, G1 5/N, G5.5/N, 2.5Y 5/1 or 5Y 5/1.

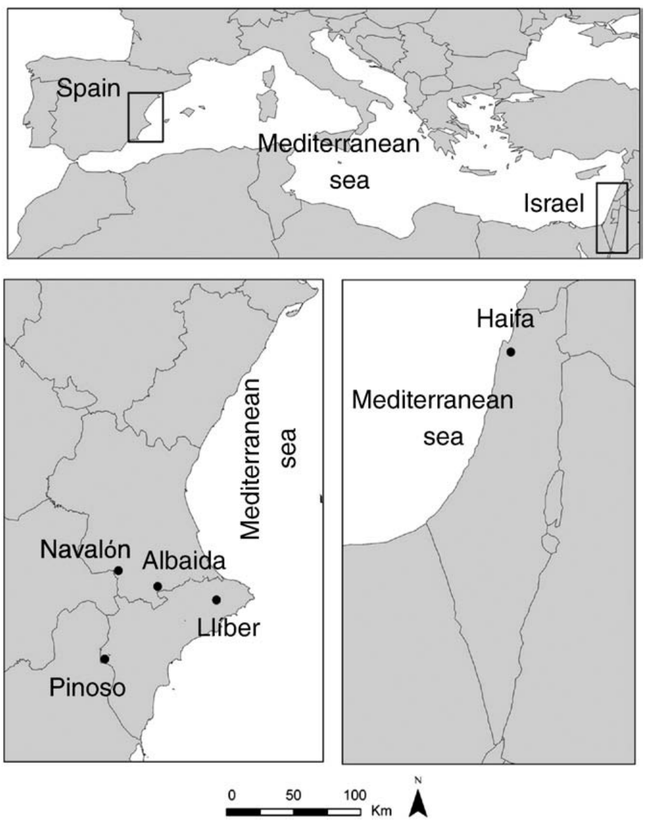

Fig. 1. Location of the study areas and sites sampled.

2.3.2. Experiment 2: preparation and measurements of ash and soil mixtures

Ash produced in the muffle furnace by heating the sampled vegetation to 250, 350, 500 and 700 °C was mixed with soil material. The mixture comprised 50 g of soil (either wettable or water repellent) plus a low (4% = 2 g) or a high (16% = 8 g) addition of ash, mixed by shaking until the mixture appeared homogeneous. The ash additions were equivalent to an ash layer thickness of ca 0.5 cm for the low dose and between 1.5 and 2 cm for the high dose, added to a soil layer of ca 3 cm. For a subsample of each composite sample, 20 mL of distilled water was added to the surface of the mixture and left to infiltrate and air-dry in the laboratory at room temperature

Table 2
Main characteristics of the soil samples used.

Soil classification (Soil Survey Staff, 2006)	Texture[a,b] (% sand, silt and clay)	SOM (%)[a]	$CaCO_3$ (%)[a]	pH[a]	EC[a] (μs cm^{-1})
Xerorthent	42, 34, 24	6.2	47.6	8.5	132

SOM: soil organic matter. EC: electrical conductivity.
[a] Values based on pooled sample material from 15 sampling points.
[b] Sand: 2–0.05 mm; silt: 0.05–0.002 mm; clay: <0.002 mm.

Table 3
Water Drop Penetration Time (WDPT) classes and class increments used in this study.
After Doerr, 1998.

	Wettable	Water repellency								
		Low			Strong			Severe		Extreme
WDPT classes	≤5	0	30	60	180	300	600	900	3600	>3600
WDPT interval (s)	≤5	6–10	11–30	31–60	61–180	181–300	301–600	601–900	901–3600	>3600

(~25 °C) for two weeks in order to explore the effect of a wetting and drying cycle on this composite material. Three replicates were made of each different treatment resulting in $n=288$ (Fig. 2). Water repellency of the composites was measured as described above.

3. Results

3.1. Water repellency (WR) of ash

Although most samples of ash from the wildfires were wettable (67%), water repellency was found in ash from all sites (Fig. 3). The frequency and persistence of WR (WDPT classes) varied between fires. The proportion of ash samples that were water repellent ranged from 13% at Navalón, where a severe fire occurred and ash samples were collected six weeks after the fire following two rain events, to 60% at Llíber, which had the lowest fire severity of the sampled sites.

Regarding ash made in the laboratory, *Q. coccifera* and *P. halepensis* ash generated in the 200 to 250 °C range was extremely water repellent, became less repellent with increasing temperature between 250 and 400 °C, and was wettable when produced above 400 °C (Fig. 4). In contrast, ash made from *R. officinalis* at 200 °C was wettable and became increasingly water repellent from 250 to 350 °C, but did not reach the severe repellence of the other two species. When produced above 400 °C, this ash type was also wettable.

3.2. Total organic carbon (TOC), colour and their relationship with water repellency

The TOC content of wildfire ash, which ranged from 4.6% to 31.1%, was generally higher than that of the laboratory-generated ash, which ranged from 22.9% to 60.3%. Wildfire samples from sites L (Llíber) and A (Albaida), which exhibited the lowest fire severity, also had the highest levels of TOC (Fig. 5). The WR of wildfire ash samples was positively and significantly correlated with TOC ($r=0.7966$, $p<0.01$) and most of the wildfire ash samples with a TOC content higher than 14% were water repellent. From Fig. 5 it is also evident that this relationship became less strong when laboratory samples are included ($r=0.5556$, $p<0.01$) as the latter generally have higher levels of TOC even when generated at higher temperatures, (TOC vs WR for laboratory samples alone produced only $r=0.3493$, $p<0.01$). For example, the highest TOC measured was from *R. officinalis* ash produced at 350 °C (60.3% TOC), which also exhibits WR, but at lower levels as for the other species studied (Table 4).

Wildfire ash samples generally became darker (more black) and more water repellent with increasing TOC content. The mean TOC contents of white, grey and black ash are 10.8 ± 4.9, 15.1 ± 5.5 and 23.8 ± 8.3% respectively, and the mean WDPT values are 1 ± 0.8, 35 ± 140 and 951 ± 1133 s, respectively. However, the relationships between Munsell colour value and TOC, and between Munsell colour value and log WDPT, are not very strong ($r=0.5702$, $p<0.01$ and $r=0.5930$, $p<0.01$, respectively) (Fig. 6). For the laboratory ash, the relationships of TOC, WR and colour are even weaker ($r=0.2257$, $p<0.01$ for Munsell value and TOC and $r=0.19922$, $p<0.01$ for Munsell value and WR). There is no gradual pattern from a lower combustion-darker colour to higher combustions-lighter colour (Fig. 7). Instead, dark and white colours are somewhat alternated.

3.3. Ash effects on soil wettability

The addition of water repellent ash at the high dose treatment caused wettable soil to become water repellent, whereas the addition of wettable ash reduced the repellency of water repellent soil (Fig. 8). The patterns are similar to those observed for ash alone. Addition of *Q. coccifera* and *P. halepensis* material burned at 250 °C and *R. officinalis* burned at 350 °C made the wettable soil water repellent but there were no changes for the other samples. These highly water repellent ash samples also enhanced the WR of the water repellent soil while all other ash samples reduced the WR of this soil, although not enough to eliminate it. However, when wetting and drying the ash–soil mixture, a reduction in WR occurred for the majority of the ash–soil mixtures, eliminating WR (WDPT<5 s) in many of the composite samples. The exceptions are the wettable soil with the most repellent ash sample, *Q. coccifera* burned at 250 °C, and the initially water repellent soil treated with the *Q. coccifera* and *P. halepensis* burned at 250 °C and *Rosmarinus officialis* at 350 °C. The higher ash dose (16% ash) had thus a strong effect on soil wettability whereas the low dose (4% ash; data not shown) resulted in very little change from the initial soil wettability for all samples. The only notable exceptions were an increase in WDPT for the wettable soil from <5 s to 146 s and to 13 s for *Q. coccifera* and *Pinus hapelensis* ash respectively, both burned at 250 °C. For the water repellent soil, WR increased 2500 s for the same previous species and

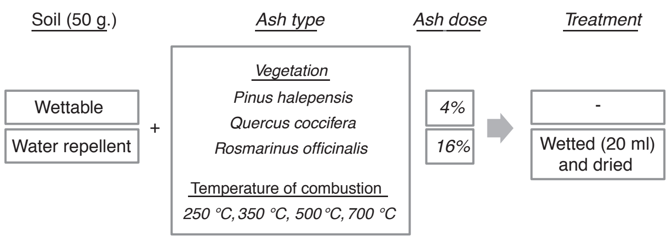

Fig. 2. Design of experiment 2 with sample numbers: soil ($n=2$), vegetation type ($n=3$), heating temperature ($n=4$), ash dose ($n=2$), wetting and drying treatment ($n=2$).

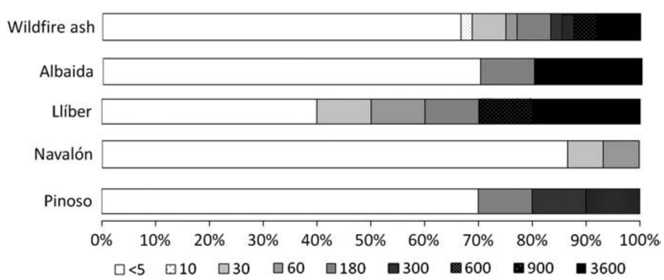

Fig. 3. Relative frequency of water repellency classes (s; see Table 3) for all wildfire ash samples combined ($n=45$) and for each wildfire individually. Data from site H (Haifa) is not included here as samples were not taken randomly, but selectively from areas with low, moderate and high fire severity.

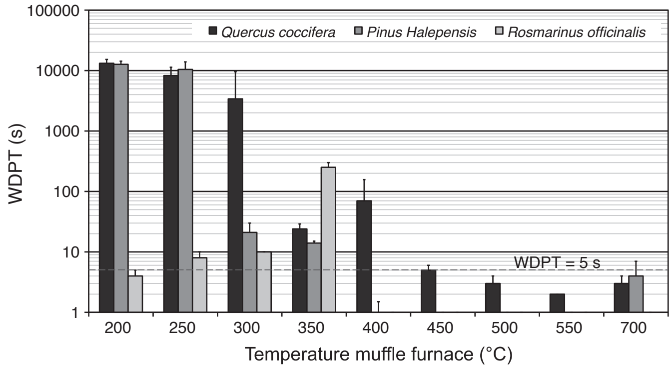

Fig. 4. Water repellency (WDPT; s) for *Quercus coccifera*, *Pinus halepensis* and *Rosmarinus officinalis* ash for each temperature used in the muffle furnace. Data represented as means ± standard deviation.

was reduced as much 2000 s in *R. officinalis* ash at 700 °C. Wetting and drying the low-dose mixture resulted in no further changes.

4. Discussion

4.1. Water repellency of ash

In previous work, ash has been considered as being highly wettable (e.g. Etiegni and Campbell, 1991; Kinner and Moody, 2010). Our data demonstrate unequivocally that ash from vegetation fires can exhibit WR (Fig. 3). Some previous studies have also indicated that ash may not be readily wettable. Khanna et al. (1996) considered the possibility that the nutrients contained in ash can dissolve only slowly (over more than 6 years) because of the hydrophobic nature of the ash amongst other factors. Gabet and Sternberg (2008) reported ash from a burn experiment using dry chaparral (*Heteromeles arbutifolia*, *Cercocarpus* and *Ceanothus*) to be very water repellent, whereas *Pinus ponderosa* ash quickly absorbed water. Finally, González-Pelayo et al. (2009) detected moderate WR for ash from a fire in *Q. coccifera, Arbutus unedo* and *P. lentiscus* vegetation in Portugal.

Wildfire- and laboratory-derived samples of the current study show that WR of ash is influenced by heating/fire severity and vegetation type. For the laboratory ash, WR decreased as burn temperature increased (Fig. 4) and *P. halepensis* and *Q. coccifera* ash had generally higher levels of WR than *R. officinalis.* For wildfire samples, WR was found to be more common and stronger where fire severity was lower, but wildfire ash will result from a mixture of different vegetation species and combustion conditions, which can be highly variable due to variations in air movement and vegetation

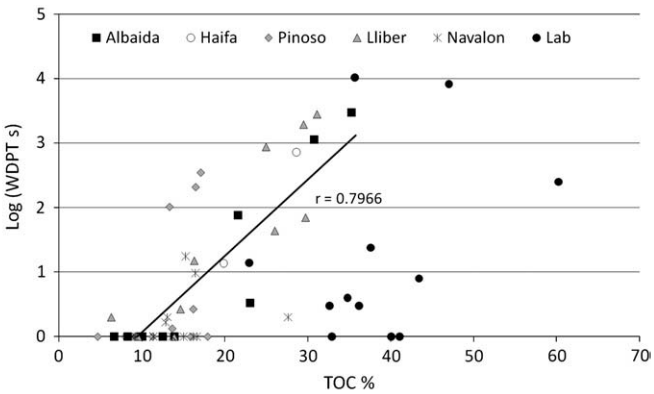

Fig. 5. TOC content (%) plotted against water repellency (log WDPT) for laboratory and wildfire samples. The r value and the linear relationship represents only wildfire samples. Laboratory samples are also plotted here for comparison.

Table 4
Mean values and standard deviations of total organic carbon (TOC% weight) for the samples created in the laboratory ($n = 3$ per sample).

Furnace temperature (°C)	Total organic carbon content (%)		
	Quercus coccifera	*Pinus halepensis*	*Rosmarinus officinalis*
Unheated control	49.6 ± 0.7	51.4 ± 0.4	56.9 ± 0.6
250	48.3 ± 0.4	39.2 ± 0.7	46.4 ± 0.6
350	40.3 ± 0.3	27.5 ± 0.8	62.0 ± 0.4
500	36.3 ± 0.8	36.6 ± 0.9	44.7 ± 1.0
700	39.7 ± 0.4	38.8 ± 0.7	44.3 ± 0.9

moisture (DeBano et al., 1998). The fact that WDPT was low at the site where it had rained prior to the sampling is consistent with the results of the wetting and drying experiment of the ash and soil mixtures. This indicates that where water repellent ash is present after the fire, its water repellency is likely to decrease following rainfall events.

4.2. Relationship of water repellency and burn temperature/fire severity with total organic carbon (TOC) and colour

Since naturally-occurring hydrophobic compounds are typically organic (Doerr et al., 2000), the reasonable correlation between TOC content and WR in ash found here is perhaps not surprising. For example, heating of soil containing organic material has been shown to increase soil WR up to a certain threshold, above which it is eliminated. Under standard atmospheric conditions, this threshold occurs broadly at around 300 °C, depending on heating duration (DeBano et al., 1976; Doerr et al., 2004), whereas under oxygen depleted (i.e. pyrolysis) conditions, WR may not be destroyed until ca

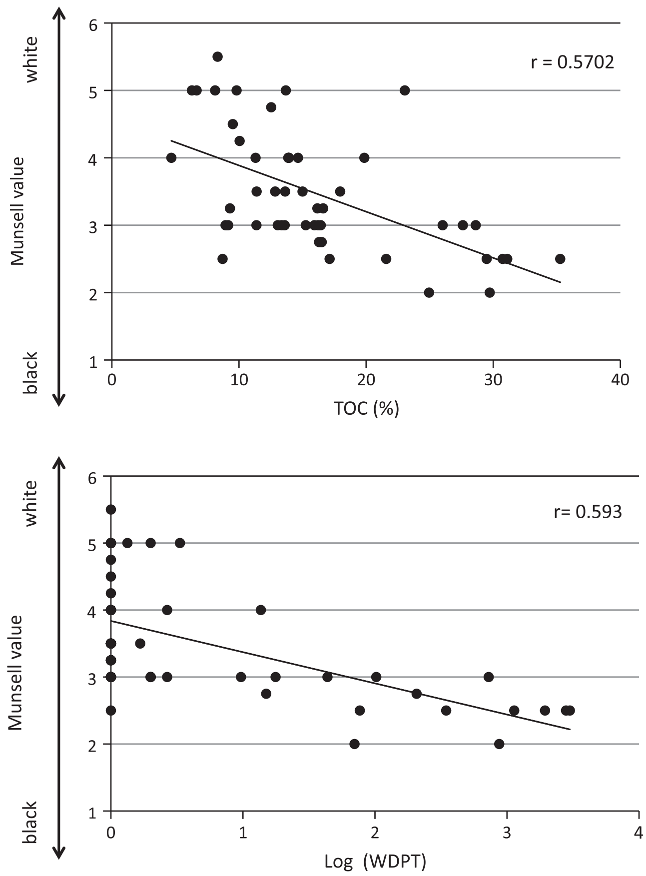

Fig. 6. Relationship between (top) colour (Munsell value) and TOC (%), and (bottom) colour (Munsell value) and water repellency (log WDPT) for all ash samples from wildfire sites.

Fig. 7. Sequence of ash colour at 200, 250, 300, 350, 400, 450, 500, 550, 700 °C (from bottom to top) of *Quercus coccifera*, *Pinus halepensis* and *Rosmarinus officinalis* burned samples (from left to right).

500 °C are reached (Bryant et al., 2005). These findings are supported by the work of Almendros et al. (1992), Baldock and Smernik (2002) and González-Vila et al. (2009), who demonstrated that thermal treatments within 250–500 °C of materials such as wood, soil and humic acids, can yield hydrophobic materials by removing external oxygen groups forms and increasing aromatic carbon amongst content. Therefore, some charred materials incorporated into soil may be sufficiently hydrophobic to increase WR in soil (Knicker, 2007). The large variety of organic compounds present in live and dead biomass that can potentially cause WR (Doerr et al., 2000) could also contribute to the variation in WR levels detected amongst all samples tested here.

The relatively few values for TOC content in ash reported in the literature vary substantially, and may reflect differences in method used and in the vegetation burned. Goforth et al. (2005) obtained rather low values for white ash (0.69 and 0.39%), and relatively low values for black ash (4.83 and 6.63%) for ash, collected, respectively, after a severe wildfire in a conifer forest and in a pine–oak woodland. For different types of straw burned in the open on large metal trays, Raison and McGarity (1980) reported ash TOC values of 2.5% and 14%. Burns (2007), for ash generated experimentally in a barrel using wood from conifer trees, measured 47–66% TOC. Philpot (1970), conducting an experiment involving pyrolysis of pine needles in a furnace, reported that at 400 °C, 35% of organic compounds remained in the ash. The values obtained here for wildfire ash match best those of Raison and McGarity (1980). The severity and hence combustion efficiency, of the wildfire studied by Goforth et al. (2005) was probably higher than for those sampled here. The values obtained by Burns (2007) and Philpot (1970) using controlled experimental conditions broadly match the ones obtained here for the ash created in a closed furnace.

Combustion efficiency (or completeness), which depends on the initial fuel moisture, the oxygen supply, quantity and quality of the fuel and temperatures reached in the fire, are thought to influence the relative proportions of black char and white mineral matter (DeBano et al., 1998). The fact that ash colour from the five different wildfires sampled here was only weakly correlated with TOC (Fig. 6) indicates that other factors may be important in determining ash colour. Hence, differences between wildfires sampled, vegetation type degree of pyrolysis of the organic matter and ash particle morphology and assemblage may all affect ash spectral properties (i.e. colour). The laboratory samples did also not follow a gradual sequence along the temperature gradient from lower combustion-darker colour-high TOC to higher combustions-lighter colour-low TOC (see Fig. 7 and Table 4), despite maximising fuel homogeneity by grinding. In contrast, Goforth et al. (2005) found a positive correlation ($r = 0.85$) between colour and TOC in ash collected for the same wildfire. Provided the vegetation distribution for their site was relatively homogenous, this would suggest that ash colour may be useful as an indicator of fire severity in homogeneously vegetated landscapes and within the same fire site. However, where this is not the case, our results suggest that ash colour may not be a very reliable indicator for comparing burn severity.

4.3. Comparison of laboratory-burned ash and field collected ash

In this study, we used wildfire ash, and laboratory ash generated ash at a specific temperature, involving broadly comparable vegetation. This provides an opportunity for examining the representativeness of ash generated in the laboratory for field conditions. The most notable result is that the TOC levels are higher for laboratory- than wildfire ash. Even the TOC values of white ash produced at 700 °C for 20 min in the furnace, which reflect relatively hot and sustained burning conditions in a wildfire, are similar or higher than for black wildfire ash. This indicates that heating experiments in the furnace do not adequately reflect burning conditions in the field regardless of the temperature used. The following factors may have also affected the process of generating laboratory ash: (i) In some cases, combustion was observed to continue for several minutes after removing samples from the muffle furnace, presumably associated with new oxygen supply. As a consequence, peak temperatures in some samples may have been different than the furnace temperature selected. (ii) Some parts of the vegetation in the crucible burned more completely than others, especially in the centre. This is indicated by the observation that in the centre ash was grey, whereas ash on the margins, top and bottom was black (Fig. 9).

Raison (1979) reported in his pioneering review that ash generated in furnaces differs quantitatively from the one produced during a wildfire. Our results support the notion that conditions in a muffle furnace are not very comparable to those of a wildfire where the rate and duration of heating are subject to rapid and drastic changes due to air movement (Gray and Dighton, 2006; Hillado, 1977). The furnace seems to restrict oxygen supply, resulting in less complete combustion compared to a wildfire for a given temperature and thus higher TOC values. Only at a furnace temperature of 700 °C, flaming combustion was observed when the sample was introduced, indicating that smoldering is perhaps more dominant in the furnace at lower temperatures, whereas outdoors, canopy leaves, for example, are mostly burned by flaming combustion while the fire advances (DeBano et al., 1998).

Finally, grinding the biomass results in a homogenisation and compaction of the fuel and minimizes the differences in mineral content between plant species (Gray and Dighton, 2006). Notwithstanding this, we still found differences in combustion efficiency between plant species for the same temperatures applied. This may be

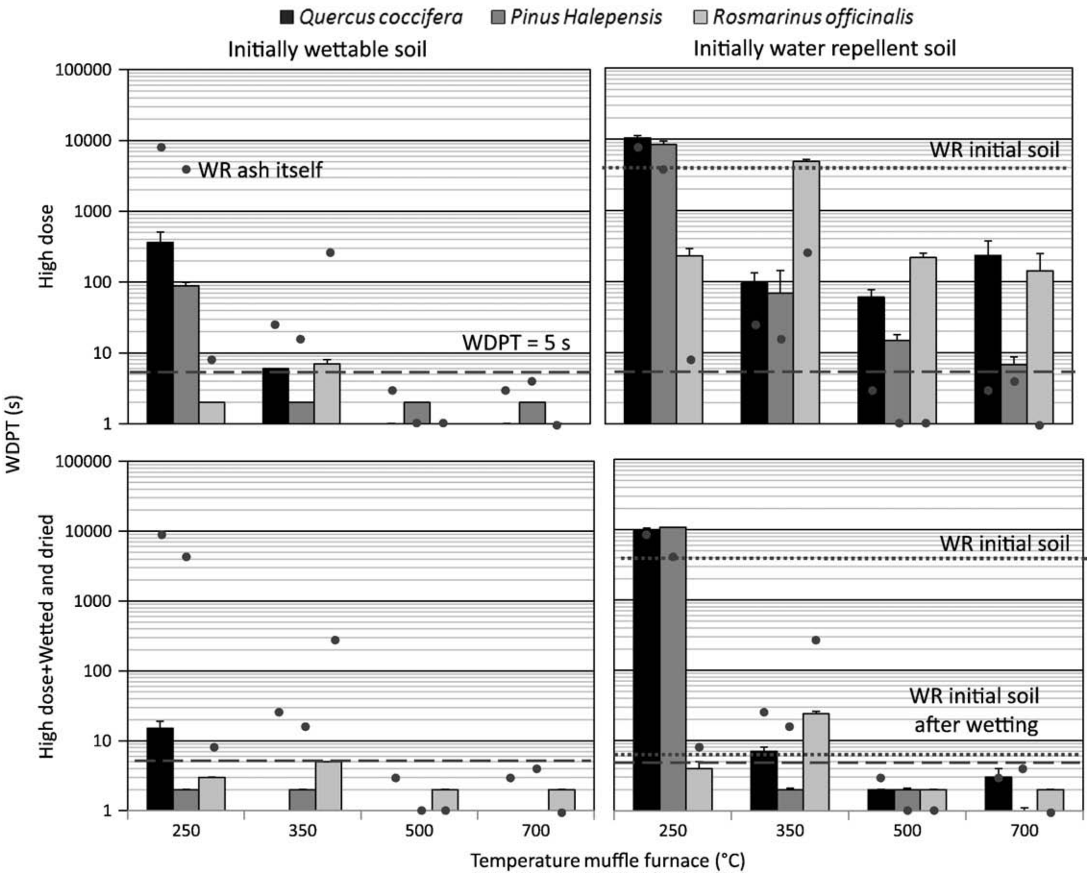

Fig. 8. Water repellency (mean WDPT in s ± standard deviation) for the soil and ash composites using *Quercus coccifera, Pinus halepensis* and *Rosmarinus officinalis* ash generated at 250, 350, 500, 700 °C and mixed with wettable (WDPT = 2 s) and water repellent soil (WDPT = 4636 s) at high dose (16%) without and with subsequent wetting treatment. The WDPT values of ash are represented by the dark dots (see Fig. 4 for details). The lower dashed line delineates WDPT of 5 s, the upper dotted line delineates the WDPT for the repellent soil (4636 s) and the lower dotted line the WDPT (6 s) for the repellent soil following wetting and drying. Data using the low dose of ash are not shown as these treatments made little change to soil WR.

due to species-dependent differences in its density, ignitability and flammability (Ormeño et al., 2009). Also Dimitrakopoulos and Papaioannou (2001) reported *P. halepensis* to be "flammable" whereas *Q. coccifera* to be "moderately flammable".

Fig. 9. Differences in ash colour within the same sample (*Rosmarinus officinalis* at 250 °C). Volatilised organic material is condensed and deposited on the crucible wall surrounding the ash.

Although it is clear that laboratory tests cannot reliably predict the response of an individual plant species during wildfire conditions, they can demonstrate differences in properties of vegetation species under certain controlled conditions, thus providing useful insight into the behaviour and impact of ash after a wildfire. To avoid potential misinterpretation of results, however, we recommend involving also wildfire ash when studying the properties of ash generated under controlled laboratory conditions.

4.4. Hydrological and ecological implications

Our results show that water repellent ash is not uncommon. Although different to soil in many respects, this suggests that the deposition of water repellent ash on the ground could have some effects comparable to those of a water repellent soil layer. Even where water repellent ash underlies wettable ash, it may form a layer hindering infiltration. However, where present, such water repellent ash is likely to be of patchy distribution. The hydrological implications of a water repellent surface soil layer, subsurface soil layer and of their patchy distribution have been examined in some detail, see reviews of Doerr et al. (2000) or Shakesby and Doerr (2006), and the effects of a water repellent ash layer may be comparable to those to some degree. However, because ash is highly erodible, any effects would be relatively short-lived. A reduced wettability of ash may also affect seed germination and vegetation growth after fire by (i) hindering water from getting to the soil surface (Mataix-Solera et al., 2007) and (ii) by delaying or inhibiting the solubilisation of the nutrients contained in the ash (Khanna et al., 1996). In contrast, where ash is readily wettable, it can reduce and delay generation of overland flow, and limit associated ash and soil erosion, due to its relatively high water storage capacity (Cerdà and Doerr, 2008; Kinner and Moody, 2007, 2010; Larsen et al., 2009; Woods and Balfour, 2008).

Our results furthermore suggest that when ash is incorporated into soil, it can increase or reduce soil WR depending on the initial respective wettability of both materials. Given that the presence of soil water repellency following wildfire can be a major concern for land managers in steep terrain due to increased runoff, erosion and debris flow risk (Shakesby and Doerr, 2006), the immediate and longer-terms effects of ash may well be of significance. Further work on the water repellency of ash and its effects on the soil, particularly under field conditions, is warranted in order to assess the wider applicability and relative importance of the findings obtained here.

5. Conclusions

This study demonstrates that ash from vegetation fires is not necessarily wettable, as widely assumed, but can be water repellent when generated under certain combustion conditions. This has direct implications for runoff, erosion and vegetation recovery after a fire. When incorporated into the soil, ash may also alter soil wettability. The results indicate that ash wettability is controlled by fuel combustion conditions and plant species. Ash generated under laboratory conditions at 200–300 °C showed the highest levels of water repellency, whereas above 400 °C, repellency was low to absent. This trend was supported by field samples, with the most repellent ash samples originating from wildfire sites with low fire severity. For the vegetation types studied here, *Q. coccifera* and *P. halepensis* litter generally produced ash with higher water repellency levels compared to *R. officinalis.* When added to soil, ash was found to substantially alter soil wettability, causing water repellency when repellent ash was added to wettable soil, but decreasing it when wettable ash was added to repellent soil. It was also found that a wetting–drying cycle can reduce the water repellency of a soil–ash mixture. Overall, the water repellency levels of ash correlated well with total organic carbon content, whereas neither of these parameters correlated very well with ash colour. This suggests that ash colour, which is widely used a parameter in classifying burn severity in the field, may in fact not be a very accurate indicator of burn severity.

Notwithstanding these findings, we suggest that experiments involving ash generated in the laboratory should be interpreted with care because combustion conditions in the laboratory may not be a very close representation of the combustion occurring in a wildfire. For example, TOC values of laboratory ash generated in our study were generally substantially higher than for ash sampled from wildfires. One reason for this difference may be the limited air flow in the laboratory, which in turn may lead to less complete combustion. We therefore suggest that studies focussing on ash from vegetation fires should also involve samples from representative outdoor fires for comparison.

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Hydrological effects of a layer of vegetation ash on underlying wettable and water repellent soil

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ABSTRACT

Hydrological processes after a wildfire may take place under soil conditions altered by heat and by the presence of ash. Soil and ash interact as a two-layer system with poorly understood hydrological properties, especially when ash covers water repellent soil. Here we quantify the effect of an ash layer (0, 5, 15 and 30 mm depth) covering wettable and water repellent soil on (i) the hydrological response and the mechanism of runoff generation and (ii) the water repellency dynamics, for a rainfall event followed by different drying periods and a second rainfall event. Laboratory rainfall simulation experiments (82.5 mm h^{-1} during 40 min) at small plot-scale (0.09 m^2) were performed and surface and subsurface flow, sediment yield, splash detachment and moisture content evolution determined. Wettable soil without ash cover generated no surface runoff, but as a two-layer system temporary surface runoff was produced when ash became saturated, until water drained through the soil. Wetting and drying changed the hydrological properties of ash, increasing surface runoff for all ash depths. Over water repellent soil, the ash layer delayed and reduced surface runoff proportionally to ash depth ($r = 0.99$), reduced soil water repellency and promoted fingered subsurface flow. Ash protected the soil from splash and sheet erosion, particularly for water repellent soil. The results demonstrate that (i) the presence of an ash layer can have contrasting effects on surface runoff, depending on the wettability of the underlying soil, and (ii) a single wetting and drying event can substantially modify ash hydrological properties.

1. Introduction

Hydrological and erosional responses of landscapes are often enhanced following a wildfire (Shakesby, 2011; Shakesby and Doerr, 2006). The causes for this change include the removal of some or all of the vegetation and litter cover (Benavides-Solorio and MacDonald, 2001; Cerdà, 1998a; Larsen et al., 2009), soil sealing (Assouline, 2004; Campbell et al. 1977; Neary et al., 1999) and heat-induced changes in soil physical properties such as the strength and spatial contiguity of water repellency (Doerr et al., 2000; Huffman et al., 2001; Mataix-Solera and Doerr, 2004; Woods et al., 2007), soil structure (Mataix-Solera et al., 2011; Neary et al., 1999), and water retention (Silva et al., 2006; Stoof et al., 2010). In addition, recent studies have highlighted that the layer of ash commonly covering the soil following fire, also plays an important role in soil wettability (Bodí et al., 2011) and the hydrological behaviour immediately after a fire (Cerdà and Doerr, 2008; Larsen et al., 2009; Woods and Balfour, 2010; Zavala et al., 2009). Post-fire hydrological processes can therefore take place under conditions of soil altered by heat, affected by the presence of a layer of ash, and no vegetation remaining to intercept rainfall or trap runoff and sediment. Before the ash is washed away by rainfall or dispersed by wind (Mataix-Solera, 1999; Reneau, 2007), the soil and ash act as a two-layer system with hydrological properties that are currently only poorly understood (Moody et al., 2009).

Ash is accepted to play a role in controlling post-fire runoff and erosion, but its effects depend on: (i) its physical and mineralogical properties (e.g. particle size, porosity, calcium carbonate content, water repellency or water retention), which vary with the temperature and conditions of combustion, and species burned (Bodí et al., 2011; Kinner and Moody, 2007; Larsen et al., 2009; Woods and Balfour, 2010); (ii) physico-chemical changes in ash after interacting with the atmosphere and water (Etiegni and Campbell, 1991); (iii) the thickness of the ash layer (Woods and Balfour, 2010); and (iv) geology and associated soil types of a given location (Larsen et al., 2009; Woods and Balfour, 2010). Ash can contribute to the temporal post-fire increase in runoff and erosion by compaction and by sealing the soil surface (Gabet and Sternberg, 2008; Mallik et al., 1984; Onda

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et al., 2008) or the reduction in runoff and soil loss by storing rainfall and protecting the underlying soil (Cerdà and Doerr, 2008; Leighton-Boyce et al., 2007; Woods and Balfour, 2008b; Zavala et al., 2009). The potential relationships between ash characteristics, soil type and ash effects, however, have remained largely unexplored. In particular, most of the studies on ash effects carried out to date have focused on wettable soil, although it is very common that an ash layer overlies soil that is water repellent. This soil property is often enhanced following a wildfire and limits the ability of a soil to absorb water, caused by the presence of hydrophobic organic compounds (DeBano, 2000). To the authors knowledge only two studies examined the effects of ash covering water repellent soil: Moody et al. (2009) and Onda et al. (2008). The hydrological behaviour of this two-layer system could be expected to differ from that with a wettable soil layer due to the reduced infiltration into the soil. In that case, the water-storage capacity of the ash layer would be even more critical. Notwithstanding this, the prolonged contact with water stored in the ash layer above, might lead to a more rapid decay of water repellency compared to a water repellent soil with no ash layer (Cerdà and Doerr, 2008; Moody et al., 2009).

To address these research gaps, we carried out rainfall simulation experiments with the aim to determine specifically the effects of an ash layer with different thickness covering wettable and water repellent soil: (i) on hydrological response and the mechanism of runoff generation and (ii) on the water repellency dynamics, for a rainfall event followed by different drying periods and second rainfall event.

2. Materials and methods

2.1. Soil and ash sampling and preparation

In order to achieve fully comparable experiments, we used identical samples of ash and soil respectively. The soil was initially wettable, but a subsample was made water repellent using the laboratory treatment described below.

The ash was collected from the Teruel wildfire (Spain) that occurred from 22nd to 27th of July 2009 in an area dominated by forest with *Pinus halepensis, Quercus ilex* and associated shrubs such as *Juniperus oxycedrus* and *Cistus* sp. The ash samples were collected a month after the fire in an area of low fire severity and before any rain or wind erosion occurred. Fire severity was estimated by the colour of the ash and other post-fire indicators described in Keeley (2009). The ash layer was thick (20–50 mm depth) facilitating the collection of a sufficient quantity of ash. The ash particle size distribution was determined by laser diffraction (Malvern Instruments Ltd., Malvern United Kingdom) and the carbon content using the loss on ignition method (Nelson and Sommers, 1996). Ash pH was determined following the procedure of Úbeda et al. (2009) by mixing a solution of ash and distilled water (1:6) for 2 h in a stirrer, which was then filtered and measured with a Crisol pH meter. Saturated hydraulic conductivity was measured using the falling head permeameter method (Klute and Dirksen, 1986) following packing of the ash samples into containers of 100 ml. This also allowed determination of dry bulk density, saturated (volumetric) water content and porosity based on initial dry weight and saturated weight (Table 1).

The soil used was collected near Southgate in the Gower peninsula, southeast Wales (UK). Soil samples were taken from 0 to 10 cm depth of the A-horizon. The vegetation cover was grass. The soil was dried and the <2 mm fraction extracted by sieving. Basic soil properties were determined for air-dried, untreated samples, which included particle size distribution (LS230 laser particle size analyzer, Beckman Coulter, Brea, CA), total organic carbon (Primacs SC-TOC automated analyzer, Skalar, Breda, the Netherlands), and soil pH (determined in a 0.001 mol L^{-1} $CaCl_2$ solution). The soil was of loamy sand texture, had a total organic carbon content of 5.5% and a pH of 5.7 (Table 1). It was fully wettable for moist and air-dry conditions with a hydraulic conductivity of 163.7 mm h^{-1}, measured once repacked with a Mini-disk Infiltrometer (Decagon Devices, Inc.). In order to obtain water repellent, but otherwise comparable soil material, a subsample was rendered water repellent by following the procedure of Urbanek et al. (2010). This involved adjusting samples to 9–18% volumetric water content, packing them immediately into autoclaving bags, equilibrated at 20 °C for 24, and autoclaving them (PS/QCS/EV150, Priorclave Ltd, London) using standard procedures (on treatment at 121 °C for 1 h) (Wolf and Skipper, 1994). The treated samples were then equilibrated again in the bags at 20 °C for 24 h. After the autoclaving treatment, water repellency had increased from 0 s (non-repellent) to > 3600 s (extremely repellent) as measured by the Water Drop Penetration Time test (WDPT; Doerr, 1998). This extreme level of water repellency limits soil wetting for prolonged periods (Leighton-Boyce et al., 2007), which facilitates isolating any effects of an ash layer on hydrological processes during rainfall events from any decay in water repellency that may occur in soil with low to moderate levels of water repellency.

2.2. Rainfall simulations

A laboratory drip-type simulator, described in more detail by Leighton-Boyce et al. (2007), was used. The simulator comprised a drip tray of 400 × 400 mm with 190 hypodermic needles of 0.5 mm diameter. Water was supplied from a 25 L tank into the drip tray. Once the water reached a certain level in the tray, a switch regulated water flowing from the cistern, minimising variation in the depth of water in the drip tray and hence rainfall intensity. An oscillating wire mesh at 500 mm below the drip tray served to break up the raindrops and ensure random drop landing positions. The intensity applied during the experiments was 82.5 ± 4.1 mm h^{-1}, resulting in 55 mm of rainfall over the 40 min of simulation. This intensity represents an intense rainstorm that has a recurrence of 6–7 years in eastern Spain where the ash had been collected. Such storms can produce substantial sediment movement and have been observed, for example, in the Valencia region in 2008 after a forest fire, where this level of intensity lasted for 10 min (Novara et al. 2011). The intensity was checked before and after each simulation by measuring the water collected in a 400 × 400 mm calibration pan placed over the plot frame for six times during a 30 s interval.

The rainfall simulation 'plot' consisted of a tailor-made metal box (Fig. 1) of 300 × 300 mm placed 1 m below the randomisation screen and with a slope angle set at 10° (17%). The box was designed to collect overland flow and subsurface flow through the soil. To achieve this, a coarse (10 mm mesh) metal screen was fixed horizontally at

Table 1
Soil and ash characteristics. The soil is soil number 3 in Urbanek et al. (2010).

Material	Soil classification[a]	Munsell colour	Texture[b] (% sand, silt and clay)	TOC[c] (%)	pH	Bulk Density (g cm^{-3})	Saturated K (mm h^{-1})
Soil	Endoleptic Cambisol	–	80, 17, 3	5.5	5.7	0.913	163.65
Ash	–	10 YR 4/1	73, 26.5, 0.5	10.36	9.42	0.30	137.75

[a] WRB: World Reference Base for Soil Resources (FAO, 2006).
[b] Sand: 2–0.05 mm; silt: 0.05-0.002 mm; clay: <0.002 mm.
[c] TOC: total organic carbon.

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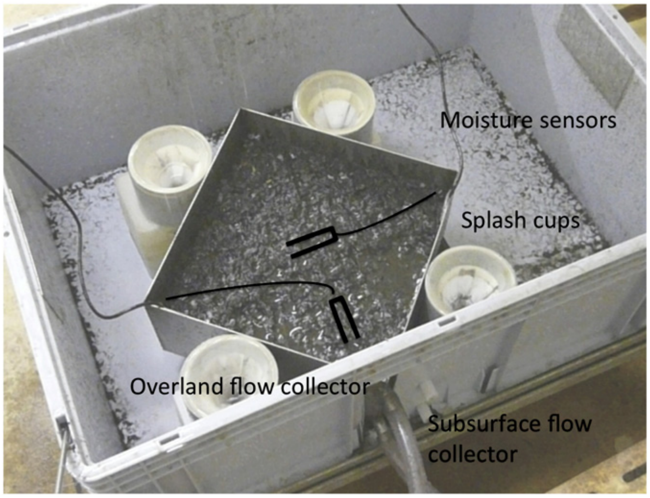

Fig. 1. Metal plot of 300 × 300 mm over which the rainfall simulation was conducted, containing an overland flow and a subsurface flow collector. Two moisture sensors were installed 15 mm below the soil surface. Four splash cups surrounded the box in order to collect sediment arising from splash detachment.

variable height in the box and covered with a cloth of permeable fibreglass. This allowed suspending a 30 mm layer of sieved soil covered with an ash layer of variable depth above the bottom of the box. Overland flow and throughflow were collected via pipes placed level with the ash surface and with the bottom of the box respectively. Two soil moisture sensors (EC-5, Decagon Services) were placed horizontally at 15 mm depth in the soil and four splash-cups (120 mm diameter) were installed next to each side of the box to allow measurement of splash-detachment for every simulation.

The rainfall simulation experiments were carried out for wettable (WDPT = 0 s) and water repellent (WDPT > 3600 s) and initially air-dry soil, for bare soil (control) and soil covered with different ash layer depths (5, 15 and 30 mm). These were aimed at simulating conditions representing the first rain event after a wildfire (Fig. 2). Three replicates per soil and ash layer depth combination were carried out. In addition, the first and second replicates were subjected to a second rainfall simulation at two different time intervals following the first simulation: (i) 24 h later, when the soil was still wet (the samples are called “2nd (a) rainfall simulation”), and (ii) 96 h (4 days) later after being partially dried in a fan assisted oven at 25 °C (samples are called “2nd (b) rainfall simulation”). The total number of simulations was 40 [1st Simulation $n = 24$ ((3 ash + 1 control) × 2 soils × 3 replicates); 2nd (a) simulation: $n = 8$ ((3 ash + 1 control) × 2 soils); 2nd (b) simulation: $n = 8$ ((3 ash + 1 control) × 2 soils)].

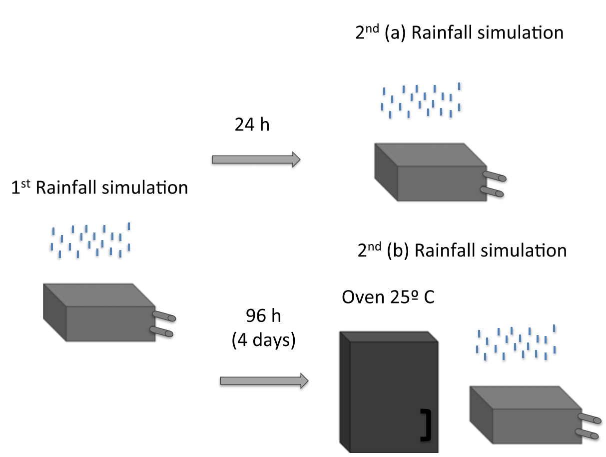

Fig. 2. Rainfall simulations time intervals. The different simulations aimed to imitate the first event of rainfall after a wildfire and a two different subsequent events: (i) 24 h after the first rainfall (the samples are called 2nd (a) rainfall simulation), and (ii) 96 h (4 days) after the first rainfall, when the ash and soil are partially dried (the samples are called 2nd (b) rainfall simulation). Bare wettable and water repellent soil and samples covered with different ash layer depth (5, 15 and 30 mm) were subjected to the first rainfall simulation and the two rainfall intervals.

2.3. Data collection

Overland flow and subsurface drainage (throughflow) were measured at 1-min intervals. The overland flow volume was collected every 5 min and stored to allow subsequent determination of sediment concentrations, sediment yield and erosion rates. The stored overland flow was passed though a vacuum filter to obtain the sediments, which were then dried 24 h at 105 °C in an oven and weighed. Soil water content at 15 mm depth, was measured every minute from the beginning to the end of the simulation.

At the end of each rainfall simulation the ash and soil profile was opened in order to examine the wetting patterns. Water repellency was measured under the ash layer, and at 1 and 2 mm soil depth. Photographs of the infiltration pattern under the ash layer and in the profile were taken. The sediment collected in the four splash cups were dried in the oven for 24 h at 105 °C and weighed.

2.4. Statistical analysis

The variables analysed were tested for normality using the Kolmogorov–Smirnov test and for homogeneity of variances the Levene test. A one-way ANOVA analysis was used to test for differences between ash depth, soil type and time interval between simulations on runoff coefficient, time to runoff, sediment yield and splash detachment. The separation of means was made according to Tukey's honestly significant difference test at an alpha level of 0.05 for all the parameters studied (variances were homogeneous). Pearson's correlation coefficients (r) were calculated to assess the relationship of ash depth with time to overland flow, time that overland flow was active, runoff coefficient and sediment yield. The analyses were performed using the SPSS Version 17 statistical software.

3. Results

3.1. Hydrological response during the 1st rainfall simulation experiment

Surface runoff coefficient was significantly higher for the water repellent soil compared to the wettable soil (20–80%; $P < 0.05$) (Fig. 3). For bare wettable soil, infiltration was 100% and the subsurface runoff (throughflow) was 57% of the total rainfall respectively, which means

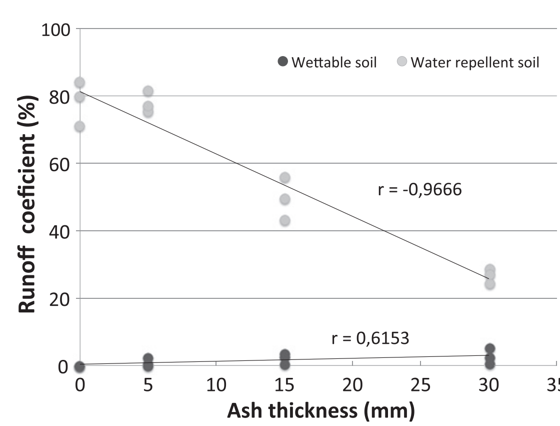

Fig. 3. Relationship (expressed as Pearson coefficient) between ash depth and runoff coefficient for a layer of ash covering wettable and water repellent soil. Three replicates per soil and ash depth were conducted.

that the soil stored 43% of the rain (2033 ± 107 ml). This was equivalent to ca 70% of the soil volume, giving a soil porosity value of ~70%. The soil covered with ash produced overland flow, reaching a maximum of 4% for an ash thickness of 30 mm. The effects of different ash thickness were not statistically significant and had a weak relationship with the surface runoff coefficient ($r = 0.6153$, $P < 0.05$; Fig. 3).

The small effects of ash thickness on soil hydrological response for an underlying wettable soil contrasted with the major, statistically significantly different ($P < 0.05$), effects for water repellent soil conditions. Surface runoff for the water repellent soil was 78% when bare and 77% with a 5 mm ash cover added. For ash thicknesses of 15 and 30 mm, the surface runoff coefficient was reduced to 50% and 26% respectively. The reduction in surface runoff compared to the bare and water repellent soil were 28% and 52% respectively. There was a strong inverse relationship between ash depth and runoff ($r = -0.9967$, $P < 0.05$; Fig. 3).

The temporal changes in surface runoff during the rainfall simulation varied with the ash thickness for either wettable or water repellent soil conditions (Fig. 4a, b). Surface runoff for the wettable soil covered by ash was ephemeral: it started later (at 4, 8 and 10 min), and lasted longer (9, 8 and 22 min) as ash thickness increased from 5 to 15 and 30 mm. Time to surface runoff was similar for water repellent and wettable soil for each ash depth, and showed a significant positive correlation with ash thickness ($r = 0.9719$, $P < 0.05$). However, for the water repellent soil, surface runoff was continuous until the end of the rainfall simulation for all ash depths, whereas for the 15 and 30 mm ash depths, surface runoff declined during the last minutes of the simulation.

3.2. Hydrological response during the 2nd rainfall simulations experiment

Changes in the visual appearance of the soil or ash surface between the first and second simulations were apparent as regards water content (a darker colour of ash and soil), surface marks of the raindrop impact and some ash redistribution in the plot downslope, but these still left the soil fully ash-covered.

For bare wettable soil, none of the second rainfall simulations, either 24 h after the first (2nd (a)), or after 96 h of drying at 25 °C (2nd (b)), produced any surface runoff. However, subsurface flow increased, especially for the rain event 24 h after the first rainfall (88% of the rain was transformed into subsurface runoff) due to the high initial moisture content (0.60 $m^3\ m^{-3}$). For the ash covered soil (Fig. 5a, b) surface runoff increased more than threefold compared to the 1st simulation and continued until the end of the rainfall. There were no statistically significant differences between surface runoff coefficient and ash depths, but significant differences ($P < 0.05$) occurred between the different second simulations, despite the fact that the shapes of runoff curves were similar. Surface runoff was 14 ± 1.4% in the 2nd (a) rainfall simulation and 17 ± 1.4% for the 2nd (b) simulation.

Regarding the water repellent soil for the 2nd (a) rainfall simulation, surface runoff was reduced for the bare soil from 75% (1st simulation) to 25% and the final infiltration rate was lower than the initial one (Fig. 6a). The soil covered with 5 mm of ash reached higher surface runoff values than for the bare soil and the three ash depths, as also happened in the 1st rainfall simulation. The experiments with 15 and 30 mm ash depth over water repellent soil showed a similar response to the ones under wettable soil during the 2nd (a) simulation, but had 10% more surface runoff and 10% less subsurface runoff

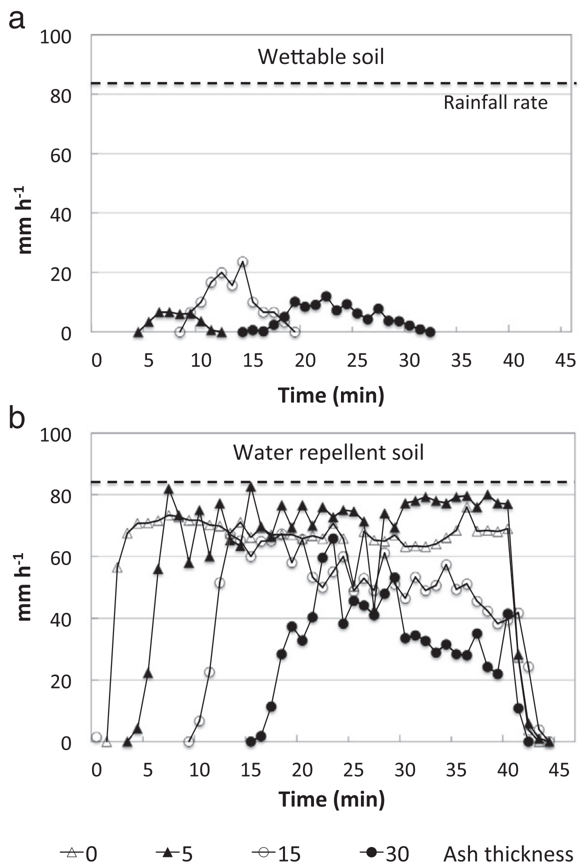

Fig. 4. Average hydrograph for rainfall simulations conducted on (a) wettable and (b) water repellent soil, for bare soil and soil covered by a layer of ash of 5, 15 and 30 mm depth for the first rainfall simulation. Each data point represents the average of the runoff rates measured for three plots. The maximum coefficient of variation was 1.73%.

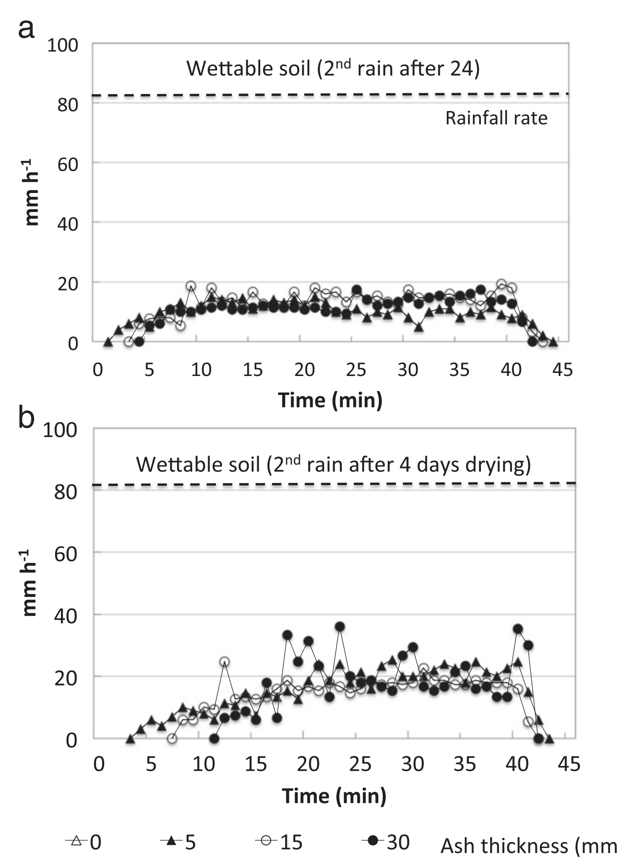

Fig. 5. Average hydrograph for rainfall simulations conducted on wettable soil bare or covered by a layer of ash of 5, 15 and 30 mm depth for (a) a second rainfall simulation 24 h after the first, and (b) a second rainfall simulation after 96 h of drying at 25 °C. Note that the simulation with no ash layer produced no runoff.

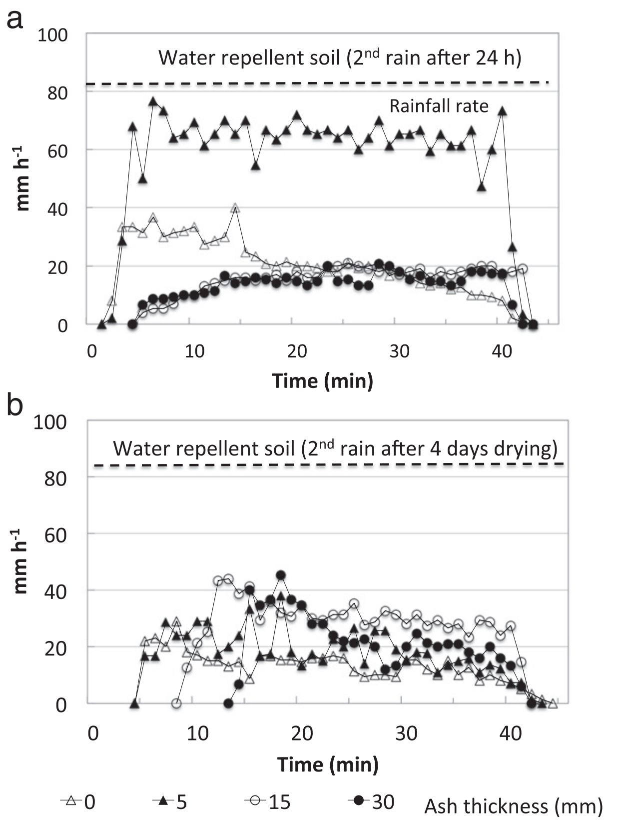

Fig. 6. Average hydrograph for rainfall simulations conducted on water repellent soil bare and covered by a layer of ash of 5, 15 and 30 mm depth for (a) a second rainfall simulation 24 h after the first, and (b) a second rainfall simulation after 96 h of drying at 25 °C.

than the wettable ones. There was no statistically significant difference ($P>0.05$) between the 15 and 30 mm ash depth for surface runoff coefficients.

The 2nd (b) rainfall simulation had a different pattern to the 2nd (a) simulation (Fig. 6b). Bare water repellent soil showed a reduction in surface runoff by 10%, and with 5 mm ash cover it was reduced by 50% compared to the 2nd (a) simulation. The experiments carried out with 15 and 30 mm ash depth, despite showing a reduction in the surface runoff of 10% compared to the 1st simulation, showed and increase in surface runoff of 10% compared to 2nd (a) simulation (Fig. 7). Final runoff coefficients were lower than the initial ones for all the treatments ($P<0.05$), i.e. bare soil and the three ash depths.

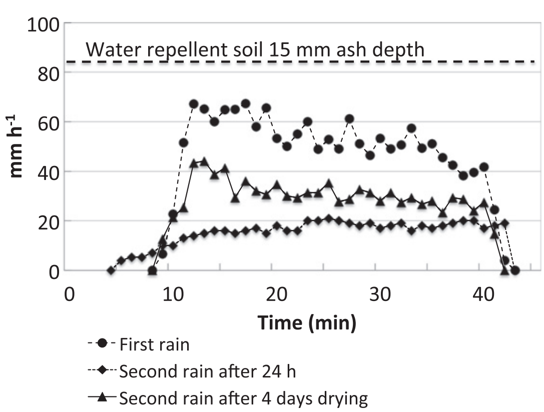

Fig. 7. Average hydrograph for rainfall simulations conducted on water repellent soil covered by a layer of ash of 15 mm depth, for the first rain, a second rainfall simulation 24 h after the first, and a second rainfall simulation after 96 h of drying at 25 °C.

Time to runoff showed no statistically significant differences ($P>0.05$) between wettable and water repellent soil for the same ash depth and same time interval between the 1st and 2nd simulation.

3.3. Soil moisture, wetting fronts and soil water repellency

Samples involving wettable soil were completely wetted after the 1st, 2nd (a) and 2nd (b) rainfall simulations, independently of the thickness of the ash layer. However, it took more time for wetting the soil as ash thickness increased (Fig. 8a). At the end of the rain, volumetric moisture reached 0.70 $m^3\ m^{-3}$, which means that soil was saturated, coinciding with the calculated water storage (rainfall − runoff).

The water repellent soil wetted to some degree, but to lower water contents compared to the wettable soil. In contrast to the patterns of the wettable soil, the moisture increased more rapidly as ash thickness increased (Fig. 8b), except for the 5 mm ash depth. For the 1st simulation, the moisture content reached a maximum of 0.37 $m^3 m^{-3}$ for the 30 mm ash depth samples. Volumetric content graphs showed a similar pattern to the subsurface runoff hydrographs, although bare soil and 5 mm of ash depth had no subsurface flow. This fact was reflected in the visual patterns of the wetting fronts: for bare soil and 5 mm ash depth soil, only the first 5 mm of the soil were wetted, while for the 15 mm and 30 mm ash depths, there was a patchy and fingered flow pattern, especially for the 30 mm ash depth samples.

Soil moisture content before the 2nd (a) simulation was slightly higher than at the end of the 1st one due to water redistribution during the 24 h between the two runs (Fig. 9). The bare surface soil was

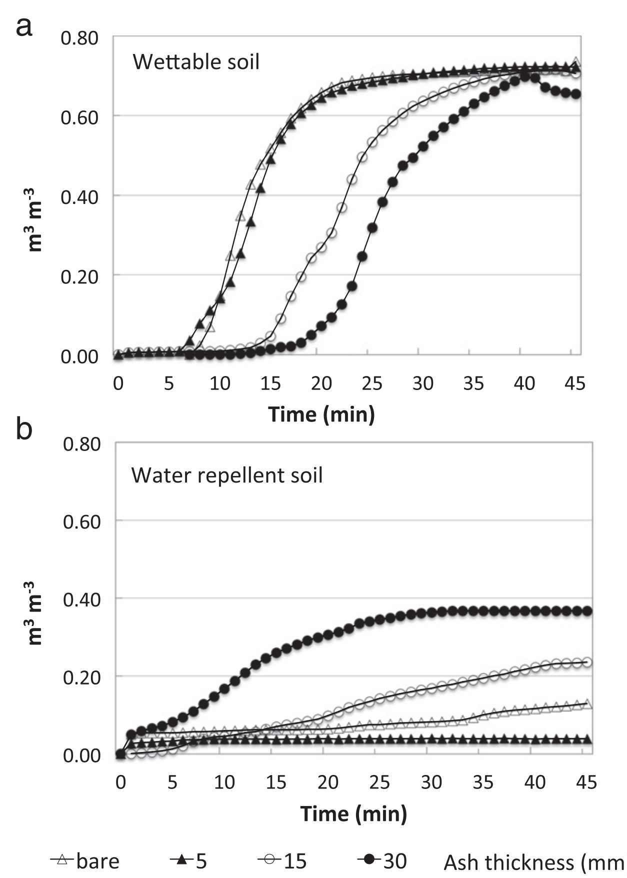

Fig. 8. Moisture content at 15 mm soil depth for (a) wettable and (b) water repellent soil, in bare soil and soil covered by a layer of ash of 5, 15 and 30 mm depth for the first rainfall simulation. Each data point represents the average of the runoff rates measured for three plots. The maximum coefficient of variation was 0.82%.

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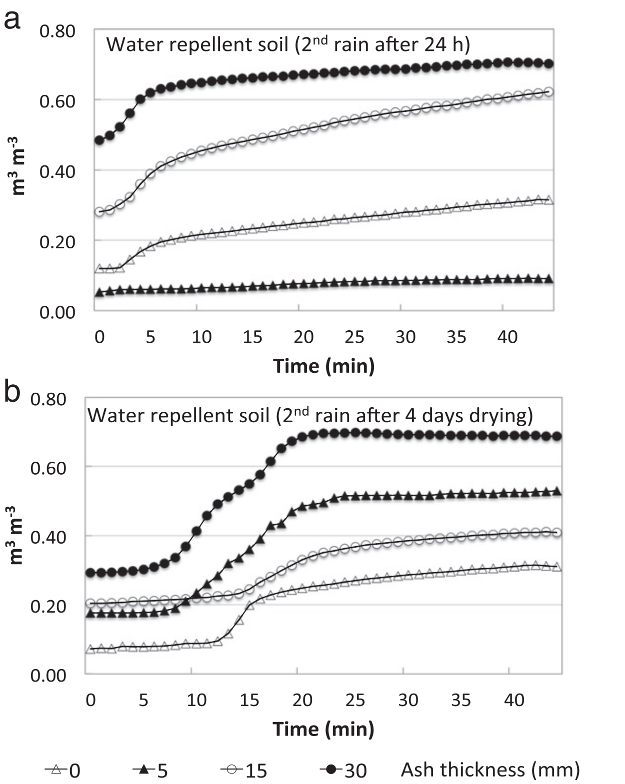

Fig. 9. Moisture content at 15 mm soil depth for wettable and water repellent soil, in bare soil and soil covered by a layer of ash of 5, 15 and 30 mm depth for (a) a second rainfall simulation 24 h after the first and for (b) a second rainfall after 96 h of drying at 25 °C.

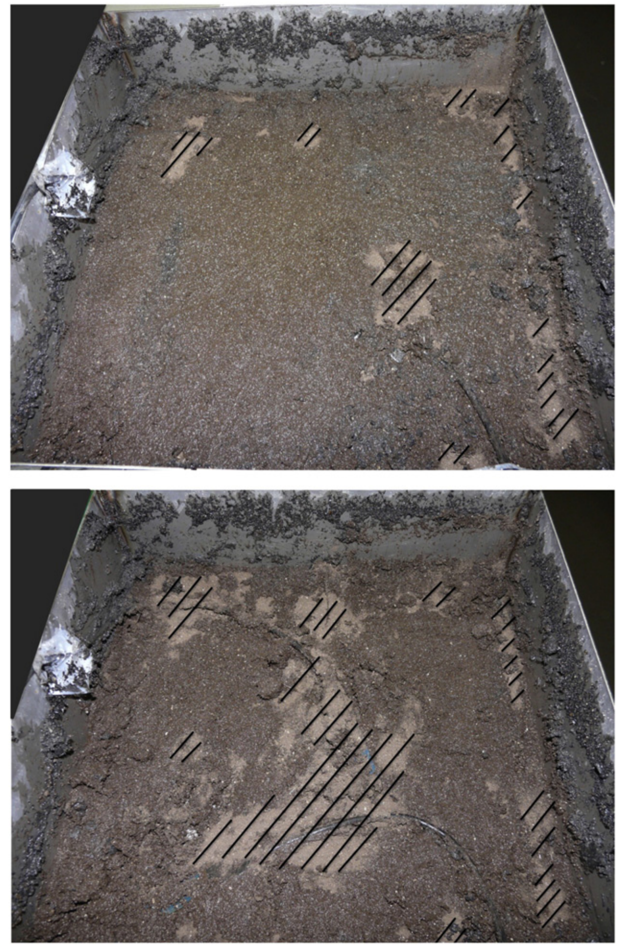

Fig. 10. Wetting pattern of the water repellent soil after the second rainfall simulation 24 h after the first, under the 15 mm ash depth (soil surface) and 15 mm depth on the soil. The areas with stripes are the dry and water repellent areas.

wet and showed no water repellency (WDPT = 0 s), however, still no subsurface flow was registered, as also happened for the 5 mm ash cover. For the 15 mm and 30 mm ash depth experiments, water flowed through the fingered wet patches (Fig. 10). At the beginning of the 2nd (b) rainfall simulation, soil moisture content was reduced during the 96 h in the oven at 25 °C (Fig. 9b) and water repellency in the surface was WDPT = 950 s. Soil moisture contents after the 2nd (b) rainfall simulation reached the same values as the 2nd (a) simulation for the ash covered samples and increased in bare soil also with the production of subsurface flow.

For all the samples water repellency had disappeared in the wet patches, but was still present with its original initial persistence (WDPT > 3600 s) in the dry zones.

3.4. Erosional response

Total sediment yield was below 6 g in all the cases for wettable soil and in the 2nd (b) rainfall simulation for water repellent soil. No statistically significant differences between the soil and ash thickness were found in any case ($P > 0.05$; Fig. 11). Sediment losses were higher only for water repellent soils during the 1st and 2nd (a) rainfall simulations.

Regarding the 1st rainfall simulation in water repellent soil, sediment yield was higher for bare soil (26.7 g) and 5 mm ash depth (24.3 g) than for 15 and 30 mm ash depth (10.5 and 9.7 g respectively). During the 2nd (a) simulation, carried out 24 h after the first, there was substantial erosion from the centre of the plot to the outlet, which uncovered the soil and subsequently led to soil material also being entrained. In this simulation, sediment yield reached 30 g and there were no statistically significant differences between ash depths and sediment yield, except for bare soil that had almost no erosion (1.4 g; $P < 0.05$). Ash and soil erosion occurred mainly at the beginning of the rain and from then on rates were nearly constant.

There was a strong positive relationship between sediment yield and surface runoff for all the samples, with the exception of the 15 and 30 mm ash depth for the 2nd (a) simulation in water repellent soil, which had the highest sediment yield for the runoff produced (Fig. 12). The Pearson correlation coefficient was $r = 0.7846$, or $r = 0.941$ ($P < 0.05$) when the exceptional samples were excluded.

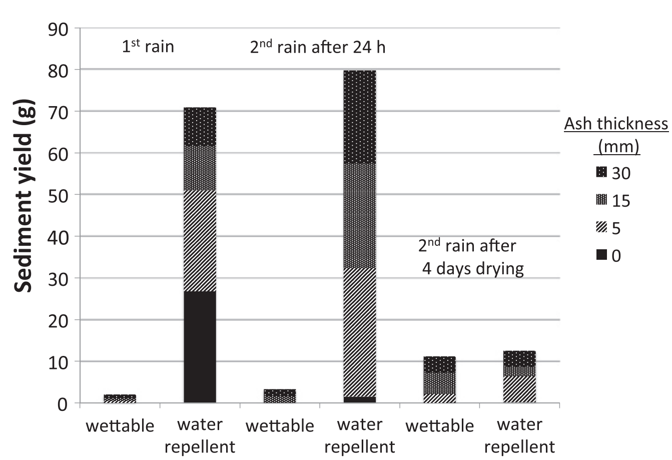

Fig. 11. Sediment yield for the wettable and water repellent soil for the first rain, a second rainfall simulation 24 h after the first, and a second rainfall simulation after 96 h of drying at 25 °C.

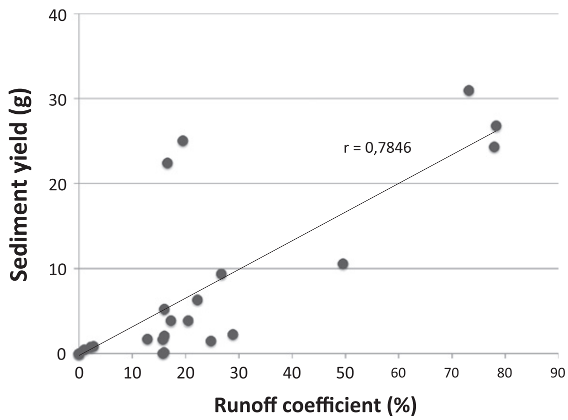

Fig. 12. Relationship (expressed as Pearson coefficient) between runoff coefficient (%) and sediment yield (g) for all the samples with different ash depth, soil type and time interval between the 1st and 2nd rainfall simulations. Pearson coefficient increases ($r=0.941$) if the 15 and 30 mm ash depth samples for the 2nd (a) simulation in water repellent soil are removed, which had the highest sediment yield for the runoff produced (two outliers in the upper left-hand side).

For these exceptional samples all the ash from the centre of the plot to the outlet had been completely removed.

3.5. Splash detachment

The total losses by splash at the end of the simulated rain events were in the order of 0.15 g on average. Statistically significant differences ($P<0.05$) were found only between bare wettable and water repellent soil for the 1st rainfall simulation. Water repellent soil samples contributed with twice as much sediment eroded by splash than the wettable soil. There were no differences ($P>0.05$) between splash detachment and ash depths for any of the scenarios.

4. Discussion

4.1. Hydrological response and underlying mechanisms for the 1st rainfall simulation

In this experiment, the bare wettable soil had 100% infiltration (applying 82.5 mm of rainfall intensity during 40 min), but when the wettable soil was covered with ash did produce surface runoff over not previously wetted ash. This observation has also been reported by Gabet and Sternberg (2008), Mallik et al. (1984), Onda et al. (2008) and Woods and Balfour (2010), although in our experiments runoff rates were smaller and ephemeral. For instance, in rainfall simulation experiments by Woods and Balfour (2010), surface runoff coefficient was 54% (average) while here 4% was the highest value.

Some of the mechanisms reported to date for ash leading to an increase in surface runoff are pore clogging or surface sealing by an ash crust (Onda et al., 2008; Woods and Balfour, 2010). Both mechanisms do not apply here for the first rainfall simulation because surface runoff stopped after some minutes. If ash sealing or pore clogging had taken place, surface runoff, once initiated, should have persisted throughout the simulation. Woods and Balfour (2010) detected pore clogging by ash even 10 months after the wildfire.

In the present study, the ash had a saturated conductivity (K) of 138 mm h^{-1}, and covered a highly permeable soil with $K=164$ mm h^{-1}. Surface runoff begun once the ash was saturated, irrespective of the water repellency of the underlying soil. Ponding at the ash–soil interface may have occurred and produced surface runoff until water drained into the soil with a higher infiltration capacity (Fig. 4a). The quantity of rainfall until surface runoff started was 8.3 ± 0.7 mm, 13.0 ± 1.1 mm and 24.2 ± 2.4 mm for ash depths of 5, 15 and 30 mm respectively. This provides an estimated ash porosity of slightly below 80%, which is consistent with published values (Bookter, 2006; Cerdà and Doerr, 2008; Woods and Balfour, 2008b; Zavala et al., 2009). Therefore, although ash saturation associated with ponding on the ash–soil interface appears to have generated some surface runoff, the general mechanism of surface runoff generation for the whole ash–soil system was infiltration-excess. This is consistent with the findings of Kinner and Moody (2010), who studied steady-state infiltration rates in a two later system consisting of soil and ash. Runoff rates are likely to differ with different rainfall intensities or soil types, especially in the case of soils with lower hydraulic conductivity. Similar mechanisms for runoff generation in two layer systems can also be found in clogged or crusted soil profiles (Assouline, 2004).

Surface runoff for simulations with water repellent soil started also when the ash had become saturated. In contrast to the experiments with wettable soil, however, it continued throughout the remaining simulation period, as the homogeneously water repellent soil could not provide any infiltration pathways. As expected, total surface runoff decreased with increasing ash depth and the associated increase in water storage capacity (Fig. 4b) (Cerdà and Doerr, 2008; Gabet and Bookter, 2011; Larsen et al., 2009; Woods and Balfour, 2010; Zavala et al., 2009). This may be associated with the increase in hydraulic pressure on the soil surface, facilitating infiltration into the water repellent soil pore space and promoting fingered wetting fronts. An exception to the improved wetting of water repellent soil in the presence of ash cover was the 5 mm ash depth (Fig. 4b). The reason might be that a 5 mm layer did not store sufficient water to exert enough pressure to facilitate soil wetting compared to the thicker layers. Given that this contrasting effect of a thin (5 mm) ash layer compared to greater ash depths was also found in Woods and Balfour (2008a, 2010) in soils clogged by ash suggests the presence of a genuine threshold effect.

4.2. Hydrological response and underlying mechanisms for the 2nd rainfall simulation

Trends were similar in the wettable soil between both of the second rainfall simulations: 24 h after the first (called 2nd (a)); and 96 h after the first (called 2nd (b)) (Fig. 5a, b). In both types of experiment, surface runoff continued until the simulation was finished.

On the basis that the bare wettable soil did not produce any surface runoff in this second simulation series, the presence of the ash layer can be considered responsible again for surface runoff generation. Visual observation suggests that at this stage raindrop impact from the first simulation had compacted the ash layer or it might be also possible that some coarse ash clogged the also repacked soil, with both processes potentially facilitated by the wetting and drying period (Etiegni and Campbell, 1991). Onda et al. (2008) reported a similar case in which overland flow over an ash layer occurred only during rain events following a first event and where the infiltration excess (Hortonian) surface runoff appeared to be dominant.

The ash compaction or ash clogging of soil pores might have occurred also for the ash covered water repellent soil, because time to runoff was the same as for the wettable soil samples for the same ash thickness for the same second rain event. However, the underlying water repellent soil controlled the overland flow production and did not allow us to distinguish whether these factors were producing any effect. In contrast, the decrease in surface runoff for the second rainfall simulations compared to the first, and overall the difference between them, was attributed to the reduction in soil water repellency, as discussed in the next section.

4.3. Ash effects on water repellency

The classic hydrograph pattern for a water repellent soil is an increase of the infiltration rate with time (DeBano, 1981) as water repellency weakens during prolonged contact with water and can be overcome with an increasing pressure head provided by ponding (Dekker et al., 2001; Wang et al., 2000; MacDonald and Huffman, 2004). In forests soils covered by litter, or burned soil with a layer or ash, it has been suggested that the storage of water in this layer may produce a prolonged water contact with the soil, which would promote the lessening of water repellency (Cerdà and Doerr, 2008; Leighton-Boyce et al., 2007).

The associated reduction in the surface runoff rates with time, an increase in the subsurface flow rates and an increase in moisture content in water repellent soil, have been observed in several cases in the present study. The clearest example occurred in the first rainfall simulation (imitating the first rain event after a wildfire) for the 15 and 30 mm ash depths (Figs. 4b and 8) compared to the bare soil and 5 mm ash depth, that might not exert enough pressure on the soil to facilitate soil wetting. As a result, while bare soil and soil with 5 mm ash depth only showed wetting of the top few mm of soil, the greater ash depths produced a patchy and fingered wetting though the soil that lead to subsurface flow. Fingered flow is the typical wetting patterns for water repellent soils (Granged et al., 2011; Ritsema and Dekker, 1994).

Regarding to the second rainfall events, the water repellent bare soil showed a reduced surface runoff rate to even lower levels than the soil covered with 15 and 30 mm ash (Fig. 6a, b). This might be due to the by then wettable topsoil layer acting similar to the ash layer in the first rain, and increased the pressure of the water and the duration of water contact. The ash layer did not contribute to further wetting in the second rainfall events due to the ash crusting or soil clogging limiting water penetration towards the soil. Therefore the ash in the second rainfall simulations did not enhance the reduction of water repellency or the soil wetting with respect to the bare soil, especially during the 2nd (b) rain.

Another important aspect is the recovery of water repellency after the drying event, since the reduction in water repellency is not necessarily permanent, as the soil may or not may become water repellent if soil moisture is reduced again (Doerr and Thomas, 2000). This was measured in the bare soil (WDPT = 0 s for the 2nd (a) rainfall simulation and WDPT = 950 s for the 2nd (b) simulation), and as indicated by the higher surface runoff rates at the beginning of the simulation, which were reduced again towards the end. This is thought to have caused the plots in the 2nd (b) rainfall simulation (after drying for 96 h) showing increased surface runoff with respect to the 2nd (a) for the 15 and 30 mm of ash depths (Fig. 7). However, the persistence of the restored water repellency (WDPT = 950 for bare soil) was not as high as the initial one (WDPT > 3600 s) and the soil wetting was still faster than in the first rain.

The fact that for all the cases water repellency in the dry soil patches remained extreme (WDPT > 3600 s) indicates that the reduction in water repellency affected its spatial extent, but not its persistence (i.e. WDPT).

4.4. Implications for soil hydrology and erosion after a wildfire

The results of this study have implications for the hydrological and erosional behaviour of soils during the post-fire period. The small-plot scale used here is particularly relevant in reflecting the functioning of the soil at pedon scale. Patches of variable soil conditions and properties (for instance different ash thickness or water repellency persistence) at pedon scale are found after a forest fire due to the heterogeneous impact of fire on soil and the not uniform ecosystem recovery (Arcenegui et al., 2007; Goforth et al., 2005; Kutiel, 1994; Neary et al., 1999). In addition, surface conditions after the first rain following a wildfire are altered not only as regards of soil moisture and consequently soil water repellency, but also ash may be crusted, eroded and redistributed (Cerdà and Doerr, 2008; Keizer et al., 2008; Onda et al., 2008)

In terms of runoff generation, this study demonstrates that a 5 mm ash depth over water repellent soil or crusted ash of variable depth can be sources of surface runoff generation and sediment, and that bare wettable soil can act as runoff sinks zone. Patches of surface runoff generation can become connected and deliver surface runoff to the channels, although the presence of sinks can reduce or prevent this from reaching the channels, whilst contributing to subsurface flow (Cerdà, 1997; Cerdà, 1998b; Imeson et al., 1992; Kinner and Moody 2010). Under field conditions, the additional presence of root holes and other structural pathways, which were excluded from our experiments, can be expected to provide additional opportunities for infiltration (Shakesby and Doerr, 2006).

The sediment yield after a wildfire is also heterogeneous in space, with areas of sediment contribution and others where sediment is accumulated (Lavee et al., 1995; Novara et al., 2011). In our study, the water repellent soil material acted as sediment source (Shakesby et al., 2000; Terry and Shakesby, 1993), and once the ash is removed by wind or surface runoff, the bare patches represent potential sites of available sediment (Moody and Martin, 2009). The greatest ash erosion was produced in the rainfall simulation 24 h after the first rain for the 15 and 30 mm ash depth over the water repellent soil. In this case, the layer of ash was removed completely from the centre of the plot to the bottom, probably due to the ash saturation with elevated water pressure and excessive amount of rain. For these two cases, the sediment yield produced was excessive in relation to the surface runoff coefficient (see two outliers in the upper-left side of Fig. 12) and indicates that incorporation of ash into the flow of runoff can increase the transport capacity. This phenomenon may be similar to the progressive debris flow associated with ash (Burns, 2007; Gabet and Sternberg, 2008). If the initial moisture content is lower, as for the 2nd (b) simulation 96 h after the 1st with the drying period, this extreme erosional event is not likely to occur.

5. Conclusions

This study has shown that the two-layer system composed of ash and soil as a result of a wildfire has different hydrological properties than ash or soil in isolation:

1. For the first rain event, ash can produce surface runoff when overlying wettable soil and it is controlled by the saturation of the ash.
2. For the first rain event, an ash layer over water repellent soil can delay and reduce surface runoff, reduce the underlying soil water repellency and promote fingered subsurface flow, proportionally to ash depth. Ash depth seems to show a critical threshold effect in terms of runoff generation.
3. Ash undergoes physico-chemical changes after the first wetting that modifies its hydrological properties and promotes the increase of surface runoff in wettable soil in the same way for all the ash depths examined. In water repellent soil the ash layer has almost no effect in the soil water repellency reduction, independently of the time interval between rainfall events.
4. The time interval between rainfall events was critical for the sediment yield production due to the changes in ash and soil moisture content, and can produce extreme erosion events that may be similar in nature to ash-laden debris flow events.
5. Overall it seems that, with the exception of extreme runoff and erosion events, ash reduces runoff and protects the soil from splash and sheet erosion, especially for water repellent soils, which are typical in post-fire landscapes and for which surface runoff and erosion is usually higher than for wettable soil conditions.

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Runoff rates, water erosion and water quality from a soil covered with different types of ash

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Key-words: White ash; black ash; rainfall simulation; water quality.

Introduction

The hydrological effects of an ash layer covering the soil after a fire are being explored. It is accepted that ash plays a role controlling post-fire runoff and erosion, but whether ash temporarily increases potential runoff and erosion by sealing the soil surface (Mallik, et al., 1984; Gabet and Sternberg, 2008; Onda et al., 2008) or reduces them by storing rainfall and protecting the underlying soil (Leighton-Boyce et al., 2007; Cerdà and Doerr, 2008; Woods and Balfour, 2008; Zavala et al., 2009) requires further investigation.

Recent studies clarify some of the reasons for these differences in observed ash effects on runoff and erosion. They are (i) the variable physical and mineralogical nature of ash depending on the temperature and condition of combustion and specie burned, e.g. size, porosity, calcium carbonate content, water repellency (Kinner and Moody, 2007; Larsen et al., 2009; Woods and Balfour, 2010; Bodí et al., 2011), (ii) changes in ash nature after interacting with the atmosphere or water (Etiegni and Campbell, 1991), (iii) the thickness of the ash layer (Woods and Balfour, 2010) and (iv) differences in geology and associated soil types of the sites (Larsen et al., 2009; Woods and Balfour, 2010). However more work is required to establish the link between ash characteristics and ash effects, particularly to enable predicting its behaviour for different ecosystems and wildfires.

In addition, differences in ash nature also can lead in differences in the chemical nature of runoff such as its nutrient content. The most common measurement method for nutrients in ash is the leaching test that involves extraction of an ash sample (Etiegni and Campbell, 1991; Pereira and Ubeda, 2010). However, this method quantifies the potential soluble nutrients in the ash, but not the actual nutrients dissolved during a rainfall event when the ash is covering the soil. Few experiments have been done on runoff quality, although the highest nutrient and suspended sediment contents in streams often occur during the first storms following a wildfire event (Hauer and Spencer, 1998; Lane et al., 2006).

To address this research gap, we quantified the effects of ash type and the thickness of the ash layer on (i) overland flow generation, and soil and ash losses, and (ii) on the chemical constituents of the overland flow generated, using a series rainfall simulations over ash-treated field plots. Differences in these parameters were also measured with time by examining the first storm after a fire and a second storm a week later.

Methods

A series of rainfall simulations were conducted in SE Spain, on an abandoned crop field with a 5º slope. We used a rainfall intensity of 50 mm h^{-1} and a duration of 60 min. The plots (0.22 m^2) were covered with two different ash types with three thicknesses: 5 mm, 15 mm and 30 mm. Control plots were also included with no ash cover. Two types of ash were used, one from *Pinus halepensis*, a grey ash collected from a low intensity wildfire, and the other a very white ash made from *Citrus sp*. litter and wood made in the laboratory at very high temperature. Ash physical and chemical characteristics (particle size, bulk density, porosity, sorptivity, particle density and total cation content) were analysed to characterise any differences between ashes. A second rainfall simulation was carried out after a week. Three replicates were made for all the treatments and a fourth replication was included to allow a destructive examination of the soil profile after the first simulation. The total number of simulations conducted was 70. Overland flow was collected at 1-min intervals and a sample retained every 10 min to allow determination of sediment concentrations, yield, erosion rates and water quality. Water samples were analysed for pH, electrical conductivity and cation content (Ca^{2+}, Mg^{2+}, K^+, Na^+).

Results and discussion

The ash from *Citrus sp*. produced at high temperature generated more runoff than the one from *Pinus halepensis* produced at lower temperatures. The effects also were different depending on the depth of the ash. A layer of 5 mm of *Citrus sp*. ash generated a runoff coefficient of 41 ± 8 %, while 15 mm produced 23 ± 15 % and 30 mm 12 ± 11 %. The runoff coefficient of the plot covered with *Pinus halepensis* ash at the same above depths was 12 ± 15%, 5 ± 5% and 5 ± 3% respectively and the average of the control plots was 6 ± 5 % (Figure 1a). Sediment yield was also higher in the white *Citrus sp*. ash than in the *Pinus halepensis* ash (figure 1b). In all cases the sediment was composed almost entirely of ash.

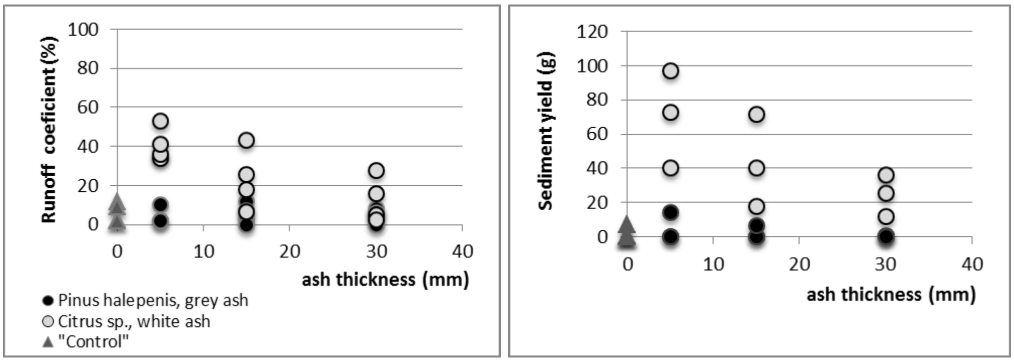

Figure 1a and 1b. Runoff coefficient (%) and sediment yield (g) of the plots covered with a layer of *Pinus halepensis* ash produced at low temperature and *Citrus sp*. ash produced at high temperature for ash depths of 5, 15 and 30 mm, for the first rainfall simulation.

For the second rainfall simulation, runoff coefficient values were reduced to 10% for *Citrus sp*. ash for the three depths and to nearly to 0% for the *Pinus halepensis* ash covered plots. Sediment yield did not exceed 20 g for *Citrus sp*. ash and was zero for *Pinus halepensis* ash.

The reason for the differences between the ash types is suggested to be due to the nature of each one. It was observed that the *Citrus sp*. ash covering the plot was crusted after the first rainfall simulation. This may be due to the hydration of this ash produced at high temperature and its high contents of calcium carbonate (Balfour and Woods, 2006).

Respect the pore clogging phenomena, Larsen et al. (2009) and Woods and Balfour (2010), studied it by examining soil thin sections or determining the porosity of the soil. They explained that clogging would occur if the fine fraction of ash is enough to clog the soil pores (if there are). The only evidence we have up to now to refuse pore clogging for this experiment, is that percentage of sample finer than 2 μm was only 5.624 ± 1.32, 0.50 ± 0.05 for the *Citrus sp*. ash and the *Pinus halepensis* ash respectively, compared with the 23% clay fraction of the soil. However, this aspect should be better studied. In addition, the lower runoff rates for *Pinus halepensis* ash can be associated to the lower bulk density, higher porosity and hence a greater capacity to hold the water. This effect increases with the thickness of the ash layer, as there will be more capacity to hold and store the water, leading to lower runoff rates (Woods and Balfour, 2010). After the second rainfall simulation, the thickness of the ash layer did not make any different effect on the water storage capacity because it become cracked and apparently more permeable.
The differences between replicates were due to variations between plots. The plots did not have the same rock fragment content and macropore distribution, and also in some of them the underlying soil exhibited slight water repellency. Nevertheless, the trends in overland flow and sediment yield within the ash types were consistent between plots.
Concerning the water quality, the runoff collected from *Citrus sp*. ash had pH values of 12 and the electrical conductivity exceeded 1000 μS cm^{-1}, whereas pH of *Pinus halepensis* was 8 and electrical conductivity around 500 μS cm^{-1}. These values were similar for the three thicknesses. The most abundant cation type in both ashes was K^+ followed by Ca^{2+}, Na^+ and Mg^{2+}. Na^+ and K^+ were higher in *Citrus sp*. ash runoff and the levels of Ca^{2+} and Mg^{2+} were similar in runoff of both ashes. However, the quantities measured here were not as high as the reported in leaching tests. Quantities reported are of the order of thousand mg L^{-1} (Etiegni and Campbell, 1991; Soto and Diaz-Fierros, 1993; Pereira and Úbeda, 2010), and here the highest value of K^+ was no more than 500 mg L^{-1}. This may be due to not all cations being washed away in the overland flow. Some would be expected them to have been lixiviated or incorporated into the soil. Also, some of them may need more water to allow full dissolution, such as Ca^{2+} and Mg^{2+} (Khanna et al. 1994).
In the second rain the quantity of cations in solution were lower, in part because the sediment concentration was lower than in the first rain, but also due to most of the cations were already washed and probably soil and ash interaction occurred during a week.

Conclusion

In this study, the different types of ash (due to either specie or temperature of combustion) and their thickness when covering the soil led to markedly different responses in overland flow, quantity of sediment washed away and water quality.
As the thickness of the ash layer increased, the water storage capacity increased as well, reducing the total runoff produced. However, in the second rainfall event, thickness did not make any difference in the overland flow, which was reduced in both cases. Regarding the type of ash, some ash types, specially the porous one, may contribute to the infiltration of water into the soil, but more dense and packed ash will increase overland flow and at the same time lead to more ash being carried within the flow. In this case, more cations contained in ash will be entrained in the runoff from burned sites.
Because of the great variety of ash types in every ecosystem with different characteristics and different underlying soil type, care should be taken in predicting ash effects on runoff rates and nutrient losses following wildfires.

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