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Review Article

# Effectiveness of post-fire soil erosion mitigation treatments: A systematic review and meta-analysis

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ABSTRACT

Wildfires are known to be one of the main causes of soil erosion and land degradation, and their impacts on ecosystems and society are expected to increase in the future due to changes in climate and land use. It is therefore vital to mitigate the increased hydrological and erosive response after wildfires to maintain the sustainability of ecosystems and protect the values at risk downstream from the fire-affected areas. Soil erosion mitigation treatments have been widely applied after wildfires but assessment of their effectiveness has been limited to local and regional-scale studies, whose conclusions may depend heavily on site-specific conditions. To overcome this limitation, a meta-analysis approach was applied to investigations of post-wildfire soil erosion mitigation treatments published in peer-reviewed journals.

A meta-analysis database was compiled that consisted of 53 and 222 pairs of treated/untreated observations on post-fire runoff and erosion, respectively, extracted from 34 publications indexed in Scopus. The overall effectiveness of mitigation treatments, expressed as the quantitative metric 'effect size', was determined for both the runoff and erosion observations, and further analyzed for four different types of treatments (cover-based, barriers, seeding, and chemical treatments). The erosion observations involving cover-based treatments were analyzed for differences in effectiveness between 3 different types of mulch materials (straw, wood-based, and hydromulch) as well as between different application rates of straw and wood materials. Finally, the erosion observations were also analyzed for the overall effectiveness of post-fire year, burn severity, rainfall amount and erosivity, and ground cover.

The meta-analysis results show that all four types of treatments significantly reduced post-fire soil erosion, but that only the cover and barrier treatments significantly reduced post-fire runoff. From the three different cover treatments, straw and wood mulches were significantly more effective in mitigating erosion than hydromulch. In addition, the effectiveness of both straw and wood mulches depended on their application rates. Straw mulching was less effective at rates below than above 200 g $m^{-2}$, while mulching with wood materials at high rates (1300 to 1750 g $m^{-2}$) produced more variable outcomes than lower rates. Results also suggest that the overall effectiveness of the treatments was greatest shortly after fire, in severely burned sites, providing or promoting the development of ground cover over 70%, and that it increased with increasing rainfall erosivity.

It can be concluded that, in overall terms, the application of the studied post-fire erosion mitigation treatments represented a better choice than doing nothing, especially in sites where erosion is high. However, the meta-analysis highlights under-representation of studies on this topic outside of the USA, Spain and Portugal. It was also observed that most of the studies were conducted at hillslope scale and tested mulching (namely straw, wood and hydromulch) and/or barriers, while larger scales and other treatments were scarcely addressed. Further efforts need to be made in testing, from field and modeling studies, combinations of existing and/or emerging erosion mitigation treatments to ensure that the most adequate measures are applied after fires.

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## 1. Introduction

Wildfires are known to be one of the main causes of soil erosion and land degradation (Shakesby, 2011), and their impacts on ecosystems and society are expected to increase in the future due to changes in climate and land use (Moritz et al., 2014). The extent of the fire-induced changes in ecosystems is based on the degree of burn severity, defined as the level of consumption of aboveground and belowground organic matter by the fire (Keeley, 2009). The consumption of vegetation and litter, and the formation of an ash layer constitute the most apparent direct effect of wildfires, reducing the protective cover of soil and, thereby, decreasing rainfall interception and surface roughness. The increased bare soil surface allows a higher transfer of kinetic energy from raindrops to the soil, thus altering its structure and favoring erosion processes (DeBano, 2000; DeBano et al., 1998; Robichaud et al., 2000). Moderate to high severity wildfires can furthermore alter soil structure by the destruction of organic matter and mineral bindings (Fernández et al., 2010; Larsen and MacDonald, 2007: Parsons et al., 2010). The infiltration capacity of soils is reduced by the breakdown of soil aggregates, decreasing the volume of pores and allowing the creation of crusts, and also by fire-enhanced soil water repellency (Doerr and Thomas, 2000; Martins et al., 2020; Malvar et al., 2016). Altogether, these changes favor runoff generation and particle detachment and thus, soil losses (Robichaud et al., 2000; Shakesby and Doerr, 2006).

Soils provide numerous and crucial ecosystem services (Weil and Brady, 2017) so their protection after wildfires is vital for maintaining the sustainability of fire-prone ecosystems. In addition, the increased soil erosion after wildfires may also have off-site consequences such as the occurrence of destructive floods and debris flows downstream from the fire-affected area. Post-fire erosion mitigation treatments mainly target the reduction of the kinetic energy of raindrops and runoff, either directly or indirectly, thereby favoring water infiltration and limiting the detachment and transport of soil particles (Cerdà and Robichaud, 2009). Post-fire erosion mitigation treatments can be classified into three major types according to their basis, i.e. an increase in protective ground cover, either directly by creating a litter layer through the application of organic residues on the ground surface ('mulching'), indirectly by enhancing vegetation recovery, or retention of runoff by creating physical barriers on hillslope and/or in streams.

The application of protective covers, mainly in the form of straw or wood-residue mulch, represent a widely used technique for post-fire soil erosion mitigation because of their cost-effectiveness (Robichaud et al., 2000, 2013a). The most commonly applied straw mulches are composed of agricultural residues from wheat, barley or rye; on the other hand, wood-residue mulches comprise a more heterogeneous set of materials, obtained from shredded or chopped tree barks, branches, and/or logs, produced in-situ or ex-situ and that may be applied in the shape of shreds, shavings, strands or chips. However, some studies have suggested that mulching may have certain drawbacks such as inhibiting vegetation recovery (Bautista et al., 1996) and, in the case of straw, introducing seeds from non-native or noxious weed species (Kruse et al., 2004). A small number of studies have also tested the use of hydromulch for post-fire erosion mitigation, despite its being costly as a result of the mixture of water, fiber mulches, suspension agents, and tackifiers (Hubbert et al., 2012; Robichaud et al., 2013b, 2013c), and in some cases the application may be complemented by seeds and chemical improvers (Prats et al., 2016b). Erosion barriers are designed to reduce runoff velocity and increase infiltration and sediment retention by shortening the length of uninterrupted flow paths (Robichaud et al., 2008a, 2008b). In post-fire environments, barriers are often made with felled burned trees that are placed along the contours to trap water and sediments. However, log debris dams have also been used as erosion barriers in burned areas, using burned trunks, and twigs from smaller trees and shrubs (Badía et al., 2015; Fernández et al., 2011), as well as straw wattles or straw bales (Kunze and Stednick, 2006; Robichaud et al., 2019). Seeding treatments in burned areas have typically been done with seeds of grasses and leguminous, either from species native to the area or well-adapted to the specific environment of the study (Badía and Martí, 2000; Robichaud et al., 2013b, 2013c). In some cases, seeding has been complemented with the use of fertilizers (Robichaud et al., 2006) or has involved seeds coated with surfactant to increase infiltration and, hence, available soil water (Hosseini et al., 2017). Agrichemicals and, in particular, flocculants such as polyacrylamides (PAM) have also been tested in recently burnt areas, as they had shown promising results in controlling erosion in agricultural fields, even at low rates of application (Inbar et al., 2015; Prats et al., 2014a and refs. therein). The application of PAM is aimed at improving soil structure to increase infiltration but especially at increasing the viscosity of overland flow and, thereby decreasing its flow velocity and capacity to detach and transport soil particles, and increasing its re-infiltration. Combinations of the above-mentioned types of treatments have also been tested in burnt areas, in particular the combination of mulching and seeding (Badía and Martí, 2000; Hosseini et al., 2017; Vega et al., 2015).

The effectiveness of treatments has been observed to vary markedly, depending on several key factors such as topography, rainfall regimes, burn severity, and time of application. For example, Prats et al. (2016b) found that differences in post-fire runoff and soil losses between treated and untreated plots were best explained by the protective soil cover that was provided by hydromulch together with the vegetation and (post-fire) litter, while rainfall intensity and antecedent soil moisture content were important additional factors to explain the temporal patterns in runoff. Robichaud et al. (2013b) also found rainfall intensity and time since fire significantly related to post-fire soil erosion in burned plots treated with several types of post-fire mitigation measures. Previous studies have highlighted the relationship between burn severity and the recovery of the ecosystem (Keeley, 2009; Fernández et al., 2013), and others have shown that, after applying post-fire mitigation treatments, vegetation recovers faster in areas burned at low severity than those burned at higher severities (Lewis et al., 2017; Larsen & MacDonald, 2007; Robichaud et al., 2013b). This suggests that in low severity burned areas vegetation recovery is fast enough that additional mitigation treatment might not be necessary, and that efforts should be focused on severely burned areas where protective ground cover is lacking. The timing of the mitigation treatments has been highlighted as a critical variable determining their effectiveness (Fernández et al., 2016a; Robichaud et al., 2013b). This follows from the well-established conceptual window-of-disturbance model, in which post-fire erosion risk declines with time-since-fire first and foremost due to vegetation recovery (Benavides-Solorio and MacDonald, 2005; Prosser and Williams, 1998; Shakesby and Doerr, 2006). The findings of Prats et al. (2016a) closely agreed with this view, with the reduction in soil losses due to forest residue mulching decreasing consistently from the first through to the fifth post-fire year (from 96 to 92, 62, 57 and 20%, respectively).

The technical reports by Robichaud et al. (2000) and Vega et al. (2013a) have exhaustively reviewed the effectiveness of post-fire soil erosion mitigation measures, with the main purpose to inform post-fire land management agencies in the USA and NW Spain. Both reports concluded that mulching was the most efficient treatment to reduce post-fire soil erosion. Outside the USA and NW Spain, however, post-fire erosion mitigation measures have not been widely or timely implemented as part of the operational response to wildfires. In Portugal, for example, funds from the EAFRD (European Agricultural Fund for Rural Development, 2019) have been used for erosion mitigation treatments following wildfires but the implementation of these treatments has rarely occurred during the first hydrological year following a wildfire (Ribeiro et al., 2020). Arguably, one of the reasons for this delayed implementation is the lack of familiarity of forest managers and planners with the array of post-fire erosion mitigation treatments that have been tested and, perhaps most importantly, with the scientific findings on their effectiveness, costs and benefits and the key factors therein. Another reason is associated to the fact that more than 90% of the

Portuguese forest land is private land (Valente et al., 2015), so forest owners tend to question the feasibility and economic viability of such techniques since their prime goal is to make the most profit from their land (Keizer et al., 2018). All things considered, a quantitative and systematic review of the full body of scientific literature on post-fire erosion mitigation treatments does not exist to date. Such a comprehensive quantitative review would not only overcome the typical limitation of local erosion studies - i.e. the important role therein of site-specific factors – but also the qualitative nature of prior, regional-scale reviews, including the abovementioned technical reports (Prats et al., 2014b; Ferreira et al., 2015; Robichaud et al., 2000; Vega et al., 2013a). Meta-analysis is now widely applied to quantify the overall effect of a treatment compared with reference conditions across a range of studies, and to test its statistical significance (Gurevitch et al., 2018). Another instance of meta-analysis applied to post-fire runoff and erosion studies is that of Vieira et al. (2015), which focused on the effect of soil burn severity determined from field rainfall simulation studies.

The overall aim of this study is to carry out a meta-analysis of the existing scientific literature on post-fire mitigation treatments, focusing on their effectiveness in reducing the runoff and soil erosion response. The specific objectives were to assess, in a quantitative and statistical manner the extent to which:

a) these treatments reduced post-fire runoff and erosion;
b) this reduction varied between the different types of treatments and, in the case of mulching, between the different types of materials and their application rates
c) this reduction varied with time post-fire, burn severity, rainfall characteristics and ground cover.

## 2. Materials and methods

### 2.1. Publication search

The Scopus database was searched in August 2020 for publications that reported on studies of the effectiveness of erosion control treatments applied after wildfires. Different combinations of the following search terms were used: fire, wildfire, erosion, sediment yields, hydrology, mitigation, treatment, recovery and prevention. From the reported studies, those that met any of the following criteria were excluded from the meta-analysis:

a) not written in English and not published in peer-reviewed journals;
b) conducted after prescribed fires, under laboratory conditions, or under simulated rainfall or runoff;
c) not involving the application of post-fire mitigation treatments, e.g. considering the natural mulch provided by needle cast;
d) not providing erosion and/or runoff measures for treated and comparable untreated areas;
e) the treatments were applied combined with different pre- or post-fire forest management techniques (i.e. salvage logging), or to mitigate their effects.

### 2.2. Data compilation, revision and gap filling

This screening resulted in a total of 34 publications (Table 1) that were analyzed for information on parameters related to site characteristics (climate, vegetation type, bedrock, soil type and texture, slope angle, aspect, ground cover), fire characteristics (date, size, and burn severity), treatment characteristics (time between fire and treatment application, type of treatment, materials used, and application rate) and runoff/erosion data collected (monitored period, technique, and contributing area). This retrieved information was used for an overall description of the meta-database.

For the meta-analysis itself, the runoff and erosion data given in the individual publications were subdivided according to study site, scale, post-fire year, type of treatment and application rate to categorize the observations. This process resulted in 598 observations, that were further screened because some of the publications partially addressed simulated rainfall and/or runoff (Robichaud et al., 2013a, 2013b), spontaneous instead of applied mulching through needle cast from scorched pine crowns (Fernández et al., 2020), or forest management operations not aimed at preventing soil erosion (Fernández and Vega, 2016b; Lucas-Borja et al., 2019). The final set comprised 222 pairs of observations (with and without treatment) for erosion and 53 pairs for runoff. For each of these observations, the average runoff coefficient (mm-runoff mm-rain$^{-1}$ year$^{-1}$) and/or the average specific erosion rate (Mg ha$^{-1}$ mm-rain$^{-1}$ year$^{-1}$) were retrieved or computed for the statistical analysis. The corresponding standard deviations were retrieved or calculated, based on the measures of variation in the publication or data provided by the authors, upon specific request. In addition, the values of rainfall erosivity, defined as the kinetic energy of raindrop impact and the rate of associated runoff, were retrieved for each of the observations from the Global Rainfall Erosivity Database (Panagos et al., 2017).

### 2.3. Statistical analysis

The effectiveness of the various erosion control measures included in the final observation sets was evaluated by means of meta-analysis (Cooper et al., 2009). This was done for all the data (referred to as 'overall' throughout the text) as well as for each of the following groups of treatments: cover treatments (mulching and mulching combined with other treatments), barriers, chemical treatments, and seeding. In addition to the effectiveness of these treatments in reducing erosion, the role of independent variables (post-fire year, burn severity, annual ground cover, and rainfall amount and erosivity) was also analyzed. All variables were divided into classes on ordinal scales. Taking into account the terminology used throughout the 34 publications, the following classes were used:

- Post-fire year: one, two, three, and four or more.
- Burn severity: low, low-moderate, moderate, moderate-high, and high. In the selected publications, different fire severity classification methodologies were followed (i.e. Parsons et al., 2010; Vega et al., 2013b), but for the purpose in the present study, the exact name of the severity class provided by the authors was retrieved and referred to as burn severity.
- Ground cover: $\leq$30, 31–60 and $>$ 60% of bare soil surface at the end of the monitoring period;
- Total annual rainfall: $\leq$500, 501–1000; 1001–1500 and $>$ 1500 mm;
- Rainfall erosivity: $\leq$500, 501–1000; 1001–1500 and $>$ 1500 MJ mm ha$^{-1}$ y$^{-1}$.

The meta-analyses in this study employed fixed effects models, which assumes that all studies in the meta-analysis share a common overall effect size with same impacts (Jain et al., 2019), and used the logarithmic response ratio or mean effect size as the test metric, as it is widely considered to be the most appropriate metric for meta-analysis of environmental data (e.g. Abalos et al., 2014; Kalies et al., 2010; Kopper et al., 2009; Vieira et al., 2015). This ratio expresses the relative difference in a response variable between a 'treatment' and a reference:

$$lnR = ln\left(\frac{\overline{x}^E}{\overline{x}^C}\right) \quad (1)$$

where $\overline{x}^E$ represents for the mean of the response variable for the treatment (E); and $\overline{x}^C$ in addition to the mean of the response variable for the reference or control (C). Besides the mean effect size, also its bias-corrected 95% confidence intervals were obtained.

The standard deviations of the runoff and erosion rates were used as weighting factors of the individual observations, referred to as

**Table 1**
General characteristics of the experiments included in the meta-analysis.

| Publication | Country | Region (site[c]) | Climate[a] | Dominant pre–fire vegetation | Burn severity[a] | Time between fire and treatment application (months) | Number of monitored years | Treatment | Material | Application rate (g m$^{-2}$) | Initial treatment cover (%) | Scale (contributing area in m$^2$) | Number of observations |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Badía and Martí, 2000 | Spain | NE – Huesca (35) | Mediterranean | *Pinus halepensis* + shrubs (*matorral*) | Moderate | 3 | 2 | Seeding | Grasses and legumes | 30 | n/a | Hillslope (8) | 8 |
| | | | | | | | | Seeding + mulching | Grasses and legumes, barley straw | 30 + 100 | 28–54 | | |
| Badía et al., 2015 | Spain | NE – Zaragoza (34) | Mediterranean | *Pinus halepensis* + shrubs (*matorral*) | Moderate – high | 24 | 2 | Barriers | Pine | n/a | n/a | Hillslope (2) | 8 |
| Bautista et al., 1996 | Spain | SE – Benidorm (37) | Semi–arid | *Pinus halepensis* + shrubs (*matorral*) | n/s | 3 | 2 | Mulching | Straw | 200 | n/s | Hillslope (16) | 1 |
| Díaz–Raviña et al., 2012 | Spain | NW – Galicia (25) | Temperate humid | *Pinus sylvestris* + shrubs (*Erica* spp., *Vaccinium myrtillus*, *Pterospartum tridentatum, Cistus* spp.) | Moderate – high | 0.25 | 0.31 | Seeding | *Secale cereale* | 10 | n/a | Hillslope (80) | 2 |
| | | | | | | | | Mulching | Wheat straw | 250 | 90 | | |
| Díaz-Raviña et al., 2018 | Spain | NW – Galicia (25) | Temperate humid | *Pinus sylvestris* + shrubs (*Erica* spp., *Vaccinium myrtillus*, *Pterospartum tridentatum, Cistus* spp.) | Moderate – high | 0.25 | 1 | Seeding | *Secale cereale* | 10 | n/a | Hillslope (80) | 2 |
| | | | | | | | | Mulching | Wheat straw | 250 | 90 | | |
| Fernández and Vega, 2014 | Spain | NW – Galicia (24) | Mediterranean with continental influence | Shrubs (*Erica australis*, *Pterospartum tridentatum, Halimium lasianthum* spp., *Halimium alyssoides)* | High | 0 | 2 | Mulching | Wheat straw | 200 | 70 | Hillslope (500) | 4 |
| | | | | | | | | Mulching | Eucalypt bark shred | 350 | 58 | | |
| Fernández and Vega, 2016a | Spain | NW – Galicia (18) | Oceanic | *Pinus pinaster* + shrubs (*Ulex europaeus*) | High | 3[b] | 3 | Barriers | Pine | n/a | n/a | Hillslope (80) | 6 |
| | | | | | | | | Mulching | Wheat straw | 150 | 58 | | |
| Fernández and Vega, 2016b | Spain | NW – Galicia (27) | Oceanic | *Pinus pinaster* + shrubs (*Ulex* sp., *Erica* sp., *Pterospartum tridentatum*) | Moderate–high | 6 | 2 | Mulching | Eucalypt bark shred | 1100 | 68 | Hillslope (80) | 1 |
| Fernández et al., 2007 | Spain | NW – Galicia (28) | Mediterranean with continental influence | *Pinus pinaster* + shrubs (*Erica cinerea, Calluna vulgaris, Pterospartum tridentatum*) | Moderate–high | 12 | 2 | Clearcutting + slash Chopping (mechanical) | Pine | n/a | n/s | Hillslope (510) | 2 |
| | | | | | | | | Clearcutting + Slash windrowing (manual) | Pine | n/a | n/s | | |
| Fernández et al., 2011 | Spain | NW – Galicia (22) | Oceanic with slight continental influence | Shrubs (*Ulex europaeus, Erica cinerea, Daboecia cantabrica, Erica arborea)* | High | 0.5 | 2 | Barriers | Cut–shrub | n/a | 6 | Hillslope (500) | 6 |
| | | | | | | | | Mulching | Straw | 250 | 80 | | |
| | | | | | | | | Mulching | Wood–shred | 400 | 45 | | |
| Fernández et al., 2016a | Spain | | n/s | | High | 1[b] | 2 | Helimulching | Straw | 250–300 | 85 | Hillslope (80) | 10 |

(*continued on next page*)

**Table 1** (*continued*)

| Publication | Country | Region (site[c]) | Climate[a] | Dominant pre–fire vegetation | Burn severity[a] | Time between fire and treatment application (months) | Number of monitored years | Treatment | Material | Application rate (g $m^{-2}$) | Initial treatment cover (%) | Scale (contributing area in $m^2$) | Number of observations |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | NW – Galicia (16, 17, 19, 23, 27) | | *Pinus pinaster* + shrubs (*Ulex europaeus)* | | | | | | | | | |
| Fernández et al., 2019a | Spain | NW – Galicia (26) | Mediterranean | *Pinus sylvestris* + shrubs (*Erica australis, Pterospartum tridentatum*) | Moderate | 0.5[a] | 2 | Mulching | Masticated pine | n/s | 43 | Hillslope (80) | 2 |
| Fernández et al., 2020 | Spain | NW – Galicia (15) | Mediterranean with oceanic influence | *Pinus pinaster* + shrubs (*Erica australis, Pterospartum tridentatum*) | Moderate | 2 | 2 | Helimulching | Wheat straw | 325 | 90 | Hillslope (80) | 2 |
| Fernández–Fernández et al., 2016 | Spain | NW – Galicia (20) | Temperate humid | Shrubs (*Cytisus striatus, Erica arborea, Ulex europaeus, Pterospartum tridentatum*) | Moderate | 1 | 1 | Mulching | Wheat straw | 80–100 | n/s | Hillslope (400) | 2 |
| Hosseini et al., 2017 | Portugal | CN (29) | Humid mesothermal | *Pinus pinaster* | Moderate | 12 | 0.83 | Seeding | Grasses | n/s | n/s | Hillslope (0.25) | 4 |
| | | | | | | | | Seeding + surfactant coat | Grasses | n/s | n/s | | |
| | | | | | | | | seeding | Pine seeds | n/s | n/s | | |
| | | | | | | | | Seeding + mulching | Pine seeds + pine needles | n/s | n/s | | |
| Hubbert et al., 2012 | USA | California (14) | Mediterranean | Shrubs (*chaparral*) | Moderate | 0.5 | 3 | Hydromulching | 40% shredded wood and 60% paper with tackifier | n/s | 50–100 | Hillslope (91.5) | 9 |
| Inbar et al., 2015 | Israel | N (38) | Mediterranean | *Pinus pinaster* | Low–moderate | 4 | 1 | PAM | Granular PAM (Superfloc A–110) | 2.5 | n/a | Hillslope (4.5) | 2 |
| | | | | | | | | PAM | | 5 | n/a | | |
| Keizer et al., 2018 | Portugal | CN (32) | Humid mesothermal | *Eucalyptus globulus* | Moderate–high | 1 | 1 | Mulching | Eucalypt bark shred | 230 | 50 | Hillslope (16) | 2 |
| | | | | | | | | Mulching | Eucalypt nark shred | 800 | 79 | | |
| Kim et al., 2008 | Korea | Gangwon–do (39) | n/s | *Pinus densiflora* + shrubs (*Quercus mongolica*) | n/s | 40 | 1.16 | Barriers | Pine | n/a | n/a | Hillslope (30) | 3 |
| | | | | | | | | Seeding | n/a | 20 | n/a | | |
| | | | | | | | | Mulching | Wood–chip | 1700 | 70 | | |
| Lucas–Borja et al., 2019 | Spain | SE – Castilla la Mancha (36) | Mediterranean semiarid | *Pinus halepensis* + shrubs (*Quercus coccifera*) | High | 2 | 0.7 | Mulching | Straw | 200 | 80 | Hillslope (200) | 2 |
| Prats et al., 2012 | Portugal | CN (31) | Humid mesothermal | *Eucalyptus globulus* | Moderate | 4 | 1.2 | Mulching | Eucalypt bark shred | 870 | 67 | Hillslope (16) | 2 |
| | | | | *Pinus pinaster* | Low | | | Mulching | Eucalypt logging slash | 1750 | 76 | | |
| Prats et al., 2014a | Portugal | CN (30) | Humid mesothermal | *Eucalyptus globulus* + *Pinus pinaster* | Moderate–high | 2 | 1 | PAM | Superfloc 110–c Series N/a–100 | 5 | n/s | Hillslope (0.28) | 2 |

(*continued on next page*)

**Table 1** (*continued*)

| Publication | Country | Region (site[c]) | Climate[a] | Dominant pre–fire vegetation | Burn severity[a] | Time between fire and treatment application (months) | Number of monitored years | Treatment | Material | Application rate (g $m^{-2}$) | Initial treatment cover (%) | Scale (contributing area in $m^2$) | Number of observations |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | Mulching | Eucalypt bark shred | 1100 | 85 | | |
| Prats et al., 2016a | Portugal | CN (30) | Humid mesothermal | *Eucalyptus globulus* | Moderate | 2 | 5 | Mulching | Eucalypt bark shred | 1100 | 77 | Hillslope (0.28–110) | 6 |
| Prats et al., 2016b | Portugal | CN (33) | Mediterranean | *Pinus pinaster* + shrubs (*Calluna vulgaris, Arbutus unedo*) | Moderate | 7 | 3 | Hydromulching | Aqueous mixture of wood Fibers, seeds, surfactant, nutrients, natural bio–stimulant, colorant | 350 | 80 | Hillslope (0.25–10) | 7 |
| Prats et al., 2019 | Portugal | CN (32) | Humid mesothermal | *Eucalyptus globulus* | Moderate–high | 1 | 2.83 | Mulching | Eucalypt bark shred | 323 | 41 | Swale (500–807) | 3 |
| Robichaud et al., 2006 | USA | Washington (3) | n/s | *Abies lasiocarpa* + shrubs (*Vaccinium scoparium*) | High | 3 | 4 | Seeding | *Triticum aestivum* | 3.4 | n/s | Hillslope (36) | 12 |
| | | | | | | | | Fertilizing | 75% ammonium nitrate/25% ammonium sulfate | 3.1 | n/s | | |
| | | | | | | | | Seeding + fertilizing | *Triticum aestivum* + 75% ammonium nitrate/25% ammonium sulfate | 3.1 + 3.4 | n/s | | |
| Robichaud et al., 2008a | USA | Washington (3), Montana (5, 6), Colorado (9), California (12,13) | n/s | *Abies grandis, Pseudotsuga menziesii, Pinus ponderosa, Pinus jeffreyi, Pinus edulis, Pinus coulteri, Quercus kelloggi* + shrubs (*Juniperus* spp., *Holodiscus discolour, Ceanothus leucodermis, Physocarpus malvaceus, Symphoricarpos albus, Artemisia* spp.) | High | 0.5[b] | 4 | Barriers | n/s | n/a | n/a | Catchment (12,000–133,000) | 4 |
| Robichaud et al., 2008b | USA | Montana (5) | n/s | *Pseudotsuga menziesii* | High | 0.5 | 3 | Contour log felling | Pine | n/a | n/a | Hillslope (100) | 9 |
| | | | | | | | | Wattles | Straw | n/a | n/a | | |
| | | | | | | | | Contour trench | n/s | n/a | n/a | | |
| Robichaud et al., 2013a | Canada | British Columbia (1) | Continental | *Pseudotsuga menziessi* + *Pinus contorta* + *Populus tremuloides* + shrubs (*Spiraea betulifolia*) | High | 2[b] | 2 | Mulching | Straw | 200 | n/s | Hillslope (84) | 4 |
| | | | | | | | | Mulching | Wood–chip | 1300 | n/s | | |
| Robichaud et al., 2013b | USA | Idaho (2, 7), Washington | n/s | *Pinus ponderosa* + *Pseudotsuga menziesii* | High | 0.5–2.5[b] | 4–7 | Hydromulching | 60% recycled paper +40% | 60 | 53 | Hillslope (23–250) | 45 |

(*continued on next page*)

**Table 1** (*continued*)

| Publication | Country | Region (site[c]) | Climate[a] | Dominant pre–fire vegetation | Burn severity[a] | Time between fire and treatment application (months) | Number of monitored years | Treatment | Material | Application rate (g $m^{-2}$) | Initial treatment cover (%) | Scale (contributing area in $m^2$) | Number of observations |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | (4), Colorado (9) | | + *Abies lasiocarpa* + *Abies grandis* + shrubs (*Juniperus communis, Arctostaphylos uva–ursi, Vaccinium scoparium, Vaccinium caespitosum, Physocarpus malvaceus*) | | | | | wood fiber mulch | | | | |
| | | | | | | | | Hydromulching | Wood fiber mulch + guar gum tackifier | 110 | 56 | | |
| | | | | | | | | Mulching | Wheat straw | 220 | 56 | | |
| | | | | | | | | Helimulching | Straw | 220–560 | 68 | | |
| | | | | | | | | Mulching | Wood strand | 450 | 54 | | |
| | | | | | | | | Seeding | Native seed mix | n/s | n/a | | |
| | | | | | | | | Mulching | Wood–shred | 1250 | 51 | | |
| Robichaud et al., 2013c | USA | California (14), Colorado (9) | n/s | *Pinus ponderosa, Pseudotsuga menziesii* + shrubs (*Juniperus communis, Arctostaphylos uva–ursi, Adenostoma fasiculatum, Ceanothus greggii*) | High | 2.5–3.5 | 5 | Mulching | Wheat straw | 220 | n/s | Catchment (15,000–52,000) | 24 |
| | | | | | | | | Hydromulching | Wood and paper fiber | 110–220 | n/s | | |
| | | | | | | | | Hydromulching | Wood fiber mulch + guar gum tackifier | 200 | n/s | | |
| Robichaud et al., 2019 | USA | Utah (10,11) | Monsoon precipitation in summer and PNW precipitation in winter | *Pinus edulis* + shrubs (*Juniperus osteosperma, Quercus gambelii*) | High | 8[b] | 2 | Straw bale check damns | Straw bale check damns | n/a | n/a | Hillslope (22–276) + catchment (2000–16,000) | 4 |
| Vega et al., 2015 | Spain | NW – Galicia (21) | Oceanic | Shrubs (*Pterospartum tridentatum, Ulex gallii, Ulex europaeus, Erica umbellata, Halimium lasianthum*) | High | 0 | 2 | Mulching + seeding | Wheat straw + seed mix | 250 + 3 | 83 | Hillslope (110) | 4 |
| | | | | | | | | Seeding | Seed Mix | 3 | n/a | | |
| Wagenbrenner et al., 2006 | USA | Colorado (8) | n/s | *Pinus ponderosa, Pinus contorta* | High | 2.5 | 4 | Mulching | Wheat straw | 220 | n/s | Swale (1400–4700) + planar hillslopes (300) | 18 |
| | | | | | | | | Seeding | Grasses | 3.4 | n/s | | |
| | | | | | | | | Contour log felling | Pine | n/s | n/s | | |

n/s: not specified.
n/a: not applicable.
[a] Denomination/classification provided by authors in the publications.
[b] Estimation based on the information provided in the publications.
[c] The number(s) provided in brackets correspond(s) to the location of the study sites represented in Fig. 2.

moderator variables. This allowed estimating the weighted-least-squares relationship between the moderator variables and the true effects (Viechtbauer, 2010). Mean effect sizes were considered to be statistically significant if the 95% confidence limits did not extend above zero (null hypothesis) and for individual effect sizes to differ significantly if there was no overlap of their confidence limits (Viechtbauer, 2010; Vieira et al., 2015). The sign of the effect size indicates if the treatment increases (+) or decreases (−) the response variable relative to untreated conditions or control (Fig. 1), with its value expressing the extent of this impact on a logarithmic scale (see e.g. Kalies et al., 2010). Since moderator variables were used, the residual heterogeneity was tested for significance using Cochran's QE-test (Cochran, 1954). This test assesses whether the variability in the observed effect sizes not accounted for by the moderator variables is larger than expected (Viechtbauer, 2014). All analyses were carried out using the R statistical package (R Development Core Team, 2012).

## 3. Results

### 3.1. Characteristics of the dataset

#### 3.1.1. General description of the observations

The dataset reveals a strong geographic bias (Fig. 2), as 56% of the 222 observations are located in the west of the USA (states of California, Colorado, Idaho, Montana, Utah and Washington) and 40% in the Iberian Peninsula, of which 28% are in Spain (regions of Galicia, Valencian Community, Aragón and Castilla la Mancha) and 12% are in Portugal (central and northern regions). Canada, Israel and Korea are the other countries represented in the dataset, with 1–2% of the observations.

The soils of the USA observations are mainly developed on igneous parent materials (granite, gabbro, granodiorite, rhyolite and basalt) and, to a lesser extent, metamorphic parent materials (schist and gneiss), while one study in the USA concerns sedimentary parent material (sandstone). The USA soils of igneous and metamorphic origin have a sandy texture, whereas those derived from sandstone have a silty loam texture. The soils of the observations in the western part of Iberian Peninsula (Galicia, Portugal) are developed on metamorphic (schist, slate and phyllite) or igneous (granite) parent materials, whilst the soils of the remaining observations in the Iberian Peninsula have a sedimentary origin (variety of calcareous materials). The former soils have either a sand or a loam texture, whereas the latter have a loamy texture. The dataset also reveals a strong bias towards the type of pre-fire dominant vegetation, with shrubs and coniferous forests being present in 77% and 68% of the observations, respectively, as opposed to eucalypt (6%) and deciduous forests (2%).

The monitoring scale of the studies ranges from micro-plots (plots smaller than 1 $m^2$), to hillslopes, swales and catchments but reveals a strong bias towards hillslope scale (Fig. 3a) with 72% of the studies as opposed to 5%, 9%, and 14% respectively. The micro-plot and hillslope observations involve a wide range of plot sizes, from less than 1 $m^2$ to 510 $m^2$. Plots of more than 100 $m^2$ (34%) are more frequent than plots between 10 $m^2$ and 100 $m^2$ (29%) and especially plots smaller than 10 $m^2$ (14%). Both the swales and the catchments also vary substantially in contributing areas, from 300 to 4700 $m^2$ and from 6900 to 133,000 $m^2$, respectively. Most of the observations correspond to sites where the post-fire mitigation treatments were applied in the first two (48%) or two to six (40%) months post-fire (Fig. 3b), while in a smaller number of cases, the treatments were applied after six to twelve months (8%) or over a year (5%). The erosion observations ($n = 222$) mainly correspond to post-fire year one (30%), two (28%) and three (20%) with only 14% of the studies having observations in post-fire year 4, and only a few observations extending into post-fire year five or beyond (Fig. 3c). Runoff observations ($n = 53$) show a similar distribution, from which 28% corresponded to the first post-fire monitored year and decrease to 2% in the seventh year. The different post-fire mitigation treatments were mainly applied, as reflected in the higher number of observations, in cases where the burn severity (Fig. 3d) was high (70%), moderate-high (10%) or moderate (19%). Regarding the distribution of observations per total measured annual rainfall, 48% correspond to <500 mm, 17% to 501–1000 mm, 18% to 1001–1500 mm, and 17% to >1500 mm (Fig. 3e). The ground cover of treated and untreated plots, expressed as a percentage of bare soil surface (Fig. 3f) was used as an indicator of the evolution of the protective ground cover, and the majority of observations were obtained from areas in which it was ≤30% (53%), followed by 31–60% (29%) and in fewer cases, >60% (18%).

#### 3.1.2. Post-fire soil erosion mitigation treatments

The total 222 observations were divided into four broad types of post-fire soil erosion mitigation treatments. The dataset has a strong bias towards ground cover as well as chemical treatments, with 149 (67%) and 11 (5%) observations, respectively, whereas only 38 observations (17%) are for barriers and 24 observations (11%) for seeding. A more detailed description of the different mitigation treatments included in each of the four categories is provided in Fig. 4 and images of the ground cover and barrier treatments identified in the meta-analysis are provided in Fig. 5.

The 149 cover treatment observations are mainly composed of straw mulch (32%), hydromulching (20%), and, to a lesser extent, wood-residue mulching (15%). The application rates vary considerably for these three types of materials, i.e. from 80 to 560 g straw $m^{-2}$ (manual and helicopter application), 60–350 g hydromulch $m^{-2}$, and 230–1750 g wood shred $m^{-2}$. The hydromulch of the bulk of the observations consists of a combination of wood fiber/shreds mulches combined with paper fiber and tackifiers (Hubbert et al., 2012; Robichaud et al., 2013b, 2013c) but also includes seeds, surfactant, fertilizer, bio-stimulant and seeds in the case of the observations from Prats et al. (2016b).

The 38 barrier observations mainly involve barriers built from plant materials, consisting of branches and twigs from burnt trees and/or shrubs (8%), of burnt tree logs felled along contour (5%), and of straw wattles or straw bales (3%). The remaining observations involve barriers in the form of soil trenches dug along contour lines (1%).

From the 24 seeding observations, 15 (7%) involve a mix of seeds typically including grass and leguminous seeds form nature species or species adapted to the environmental conditions of the burned ecosystems. The remaining 9 (4%) observations involve seeding with either rye (*Secale cereale*) or wheat (*Triticum aestivum*). The seeding application rates reported by the authors vary from 3 to 30 g $m^{-2}$, and in the case of one publication, it is of 270 seeds $ha^{-1}$ (Robichaud et al., 2013b).

The final 11 observations involve the use of chemicals i.e. polyacrylamide (PAM) and fertilizers. In three observations, PAM was applied at the soil surface in granular form at rates of 2.5–5 g $m^{-2}$. The eight fertilizer observations, four of which in combination with seeding, involve a mixture of 75% ammonium nitrate and 25% ammonium sulphate that was applied at a rate of 3.4 g $m^{-2}$.

#### 3.1.3. Post-fire runoff and erosion rates

The runoff volumes and erosion rates reveal a clear overall tendency towards lower values in treated areas compared to those without the emergency stabilization treatments, with 72% of the 53 runoff observations and 81% of the 222 erosion observations plotting above the 1:1 line (Fig. 6a and b).

The untreated observations indicate a wide range of runoff from 0 to 95 mm and erosion values from 0 to 55 Mg $ha^{-1}$. The importance of post-fire erosion is well illustrated by the fact that 52% of the untreated values exceed the precautionary threshold range of tolerable hillslope soil erosion of 0.3 to 1.4 Mg $ha^{-1}$ $y^{-1}$ proposed by Verheijen et al. (2009) for Europe. Furthermore, the 90th percentile of untreated values of 22 Mg $ha^{-1}$ demonstrate the sometimes extreme nature of post-fire erosion.

### 3.2. Overall effect size of post-fire mitigation treatments on runoff

The various erosion mitigation treatments together have a highly significant overall effect on runoff (ES = −0.50, $p < 0.001$; Fig. 7), on average reducing it by 52%. Also the cover treatments significantly reduce runoff (ES = −0.44) and so do the barrier treatments together (ES = −0.82), but the levels of significance are clearly lower ($p < 0.05$), related to the variability in results. The seeding and chemical treatments could not be statistically analyzed due to their reduced number of observations but are shown for the sake of completeness (Fig. 7). Further analyses on the effectiveness of mitigation treatments at reducing runoff could not be conducted because of the limited number of runoff observations.

### 3.3. Effect size of post-fire mitigation treatments on soil erosion

#### 3.3.1. Overall effect size

All erosion treatments considered together also produce a highly significant reduction in post-fire soil erosion (ES = −1.10, $p < 0.001$; Fig. 8a). The mean effect size corresponds to an average decrease in erosion of 56%, indicating a slightly greater effectiveness than in the case of runoff. Also, separately, the four main types of treatments have a significant effect on erosion. The mean effect sizes suggest a clear trend in mitigation effectiveness, decreasing from the cover treatments (ES = −1.28, $p < 0.001$), to the seeding and barrier treatments (ES = −0.81 and − 0.81, $p < 0.001$ and 0.01) and, ultimately, the chemical treatments (ES = −0.44, $p < 0.001$). Even so, only the mulching and chemical treatments differ significantly in effect size, also reflecting the relatively large confidence intervals for the barrier and seeding treatments.

#### 3.3.2. Effect size of cover treatments

The large number of observations, as well as their balanced distribution among the three main types of materials that were applied to provide protective cover to burned soils (i.e. straw, hydromulch and wood mulch), allowed separate meta-analyses to be conducted (Fig. 8b). The observations involving straw mulching with and without seeding were analyzed together, and so were the observation involving wood-residue mulching, slash chopping/windrowing and mulching by in-situ tree mastication. Mulching with all three types of materials has a highly significant effect on post-fire soil erosion ($p < 0.001$). However, hydromulching is significantly less effective in reducing erosion than mulching with straw or wood materials, reflecting a strong contrast in mean effect sizes of −0.52 as opposed to −1.66 and − 1.50, respectively.

#### 3.3.3. Effect size of application rates of straw and wood mulches

The high number of mulching observations also allowed meta-analyzing of the role of application rate, not only for straw but also for wood mulches. In both cases, three classes of application rates were distinguished and all of them have a significant effect on post-fire erosion rates, with ES ranging from −1.01 to −2.16 and the corresponding $p$'s from <0.001 to <0.01 (Fig. 8c and d). In the case of straw mulch, the lowest application rates (<200 g $m^{-2}$) are significantly less effective at reducing erosion than the higher application rates, while application rates exceeding 250 g $m^{-2}$ are not significantly more effective than the intermediate application rates of 201–250 g $m^{-2}$. The wood mulches show a similar pattern, as effectiveness is lowest for the lowest application rates. The differences in effect size between the three wood mulch classes, however, are not significant. This partly reflects relatively large confidence intervals for mulching with wood materials at rates up to 450 g $m^{-2}$ and especially rates exceeding 1300 g $m^{-2}$.

### 3.4. Effect size of ancillary variables in post-fire soil erosion mitigation effectiveness

The effect size of the post-fire treatments was calculated, separately, against four ancillary key-variables in the post-fire erosive response, post-fire year, burn severity, ground cover and rainfall.

#### 3.4.1. Post-fire year

All mitigation treatments considered together have a significant overall effect on post-fire erosion during each of the post-fire years ($p <$ 0.001, Fig. 9a). The mean effect sizes reveal some tendency to decrease with time-since-fire but only for the first three post-fire years (ES = −1.33, −1.05, −0.94, respectively) without any of these differences being significant. The 95% confidence intervals also reveal a temporal pattern, suggesting that the treatments' effectiveness becomes increasingly variable with years post-fire. The post-fire year approach was considered the best approximation for studying the effect of time, because the ES results for the timing of treatment application are inconclusive.

#### 3.4.2. Burn severity

The various mitigation treatments as a whole produce a significant reduction in soil erosion for all three classes of burn severity (high, moderate-high, and moderate) that are adequately represented in the data set ($p < 0.001$; Fig. 9b). These three severity classes show no relation of mean effect size with burn severity, as the effect size increases from moderate to moderate-high severity but then decreases again from moderate-high to high severity (ES = −1.14, −1.58 and − 1.00,

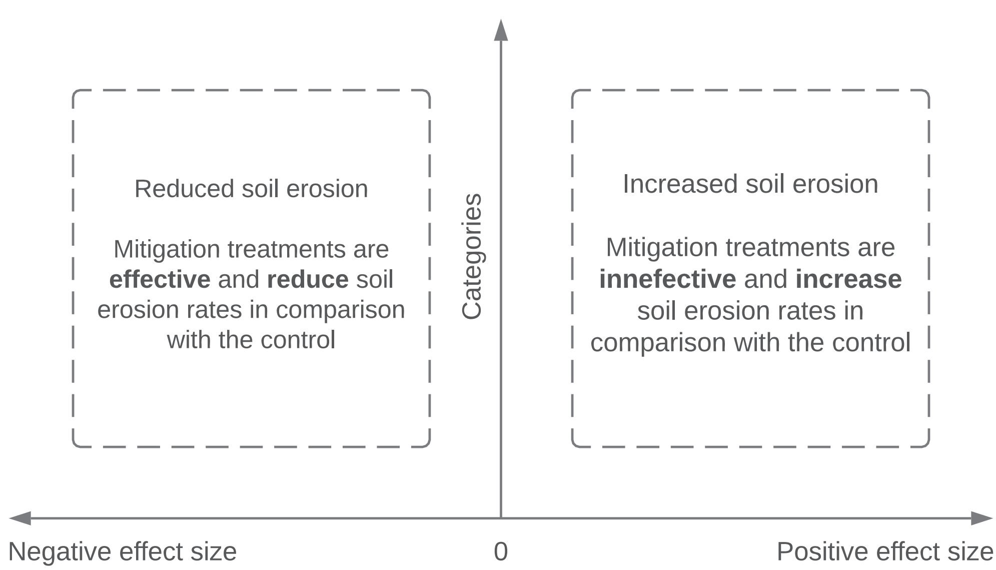


**Fig. 1.** Interpretation of the effect size analysis in relation to the effectiveness of post-fire soil erosion mitigation treatments.

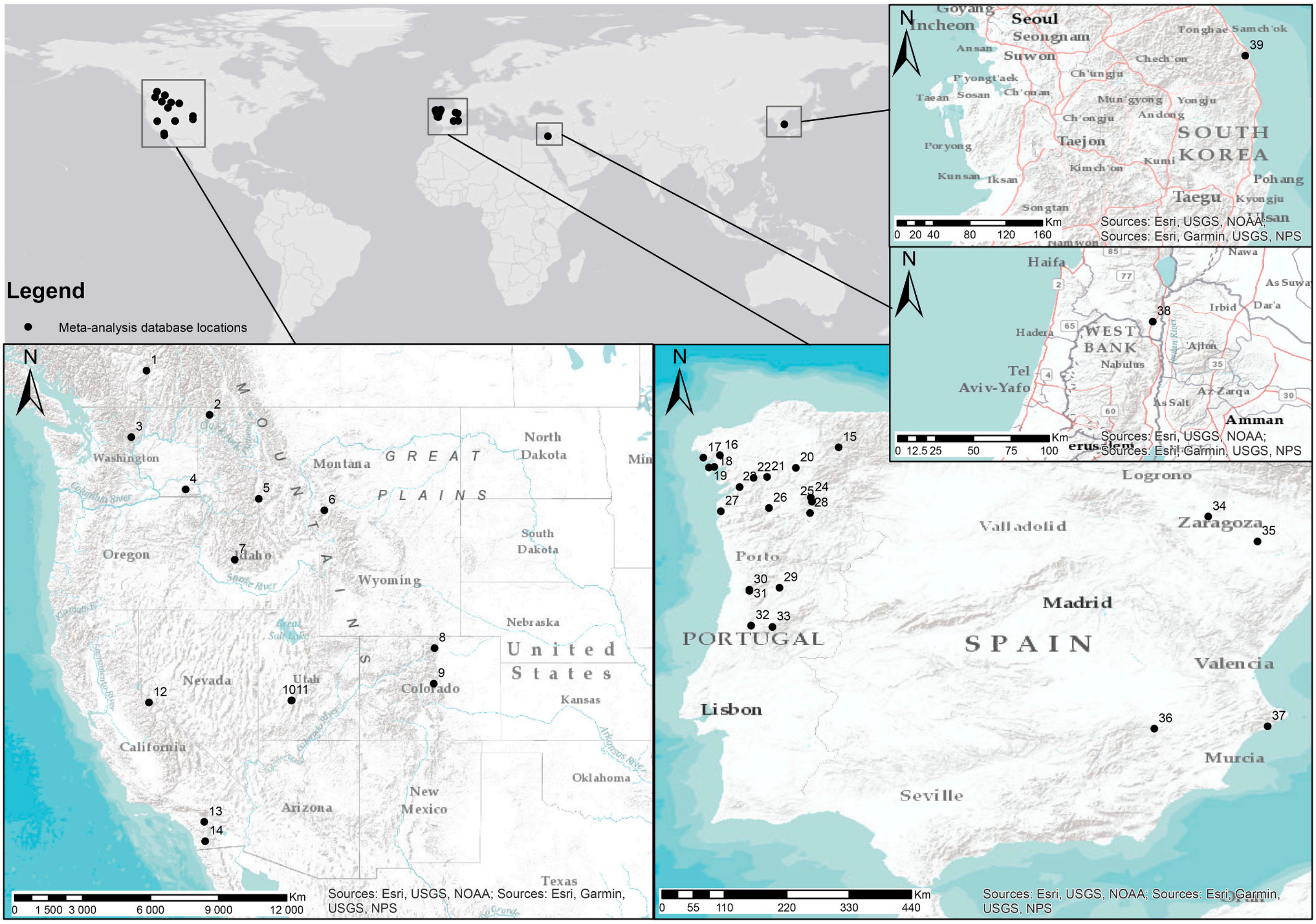


**Fig. 2.** Location of the study sites identified across the publications included in the meta-analysis. Numbers correspond to the study site reference provided in Table 1.

respectively). No differences in mean effect size between severity classes are significant.

#### 3.4.3. *Post-fire ground cover*

All mitigation treatments lead to a significant reduction in post-fire erosion (Fig. 9c), independent of whether the bare soil surface (following the implementation of the treatment) is smaller than 30% ($p < 0.001$), larger than 60% ($p < 0.01$) or in between ($p < 0.001$). The mean effect size, however, is significantly higher for the lowest cover class than for the other two cover classes (ES = −1.54 vs. -0.78 and −0.47). Likewise, it is also higher for the intermediate cover class than for the highest cover class, but this difference is not statistically significant.

#### 3.4.4. *Rainfall*

All of the mitigation treatments have a significant impact on post-fire erosion for the four annual rainfall classes distinguished in this study ($p < 0.001$, Fig. 10a). The mean effect sizes do not vary in a simple manner with rainfall amounts, first decreasing from ≤500 to 501–1000 mm, then increasing from 501 to 1000 to 1001–1500 mm and, finally, decreasing again from 1001 to 1500 to >1500 mm (ES = −1.08, −0.62, −1.61, −1.18, respectively). By contrast, the 95% confidence intervals increase with increasing rainfall amounts. The mitigation treatments are significantly less effective at reducing post-fire erosion when annual rainfall amounts to 501–1000 mm than when it amounts to either ≤500 mm or 1000–1500 mm. As an alternative to such metrics, an additional effect size analysis was performed considering rainfall erosivity as the explanatory variable (Fig. 10b). The set of post-fire mitigation treatments have a highly significant effect ($p < 0.001$) in reducing soil erosion for all the rainfall erosivity classes. This analysis also shows a general trend towards increasing mitigation efficiency with increasing rainfall erosivity (ES = −1.01, −0.97, −1.27, −1.36, respectively).

## 4. Discussion

Several reviews have explored the key variables driving the hydrological and erosive response after fires (Shakesby and Doerr, 2006; Shakesby, 2011), described the different post-fire mitigation treatments (Ferreira et al., 2015), or quantified the efficiency of mulch protective covers in diverse environments (Prosdocimi et al., 2016). However, this meta-analysis is the first quantitative synthesis of the published literature on the effectiveness of post-fire mitigation measures at reducing runoff and soil erosion under field conditions.

### 4.1. *Limitations of the current meta-analysis*

The distribution of studies identified in this meta-analysis testing post-fire runoff and soil erosion mitigation techniques revealed a strong geographical bias. Such a distribution limited the variety of studied environments, climates, and possibly reflected national funding priorities and/or existing research teams dedicated to post-fire management. The usage of a normalized indicator of runoff and soil erosion that considers the rainfall amounts allowed us to compare these observations globally. However, aspects such as climate seasonality or rainfall intensity, which are important predictors of post-fire runoff and soil

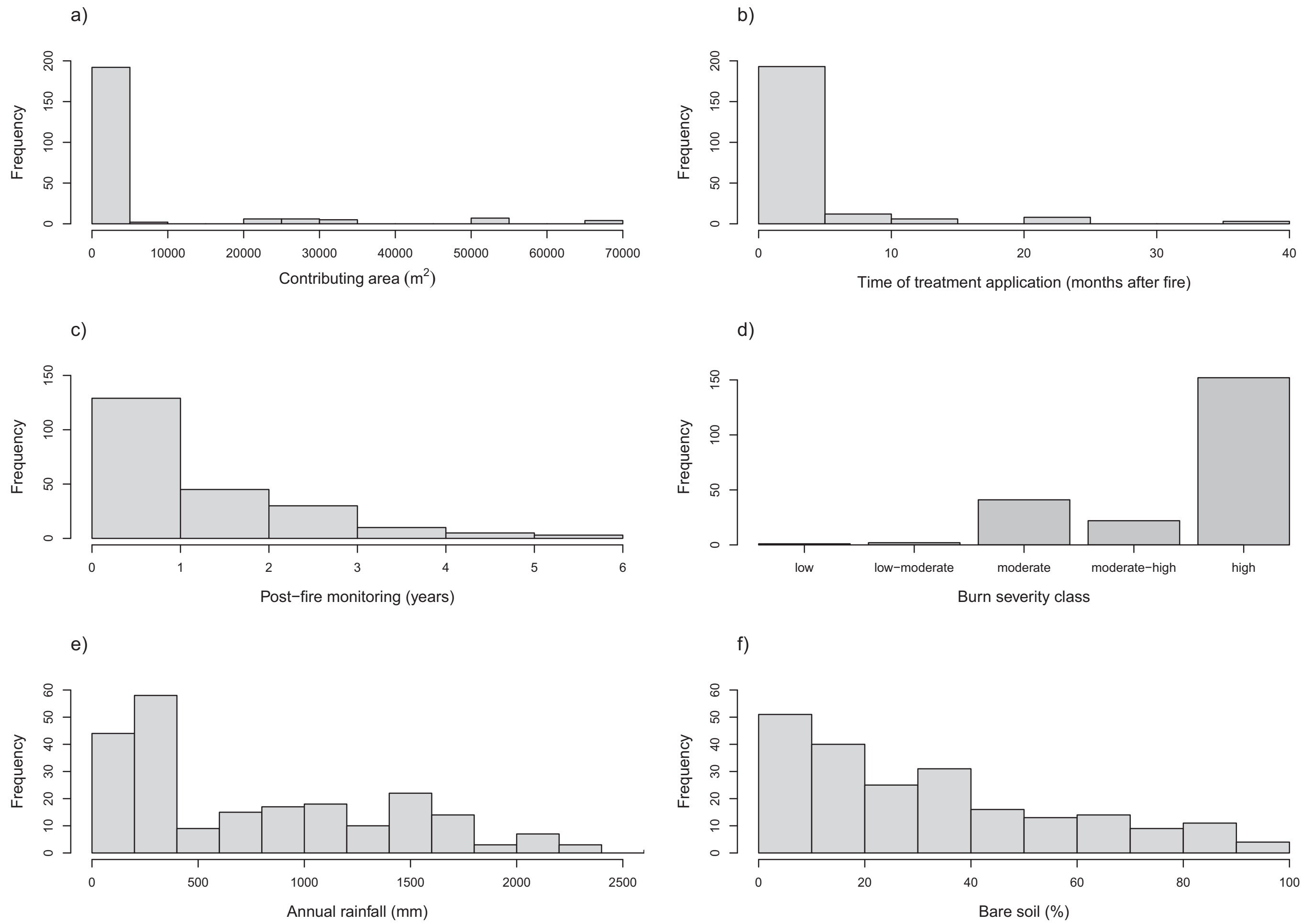


**Fig. 3.** Histograms showing the value distribution of the observations for: a) contributing area of the erosion measurement plots; b) elapsed time between fire occurrence and treatment application; c) post-fire monitored years; d) burn severity class; e) total annual rainfall and f) percentage of bare soil surface.

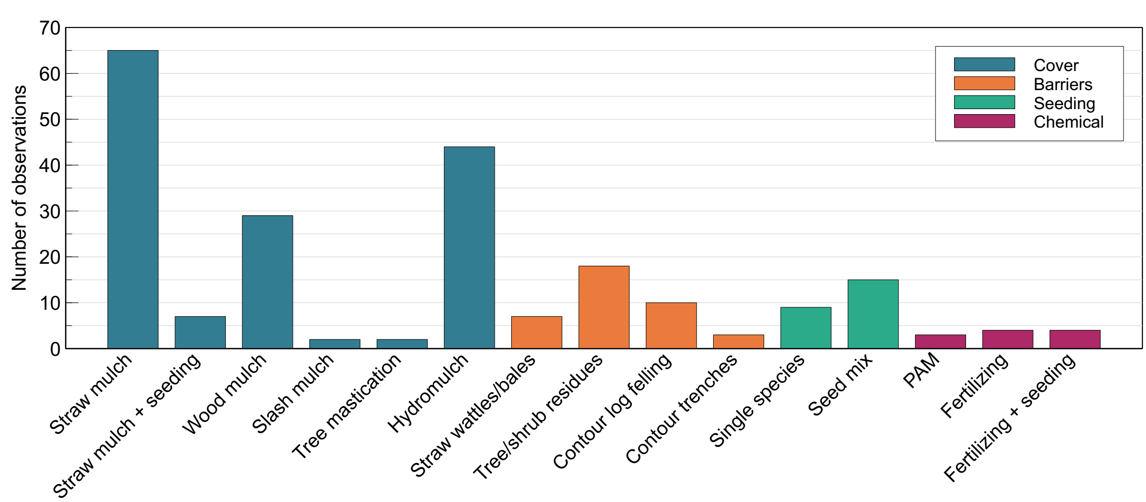


**Fig. 4.** Number of observations (pairs of treated vs untreated values) for the four major types of post-fire erosion mitigation treatments: cover treatments ($n$ = 149), barriers ($n$ = 38), seeding ($n$ = 24) and chemical ($n$ = 11). PAM: polyacrylamide.

erosion (Prats et al., 2016a, 2016b, Vieira et al., 2018), could not be considered in this analysis because of the heterogeneity of methodologies across the various studies and/or the lack of information provided in the publications. This geographical bias may easily limit the validity of the conclusions of this study for the measures' effectiveness in burnt areas in other regions, especially where the principal drivers of post-fire runoff and erosion are distinct. This includes regions that are well documented to be prone to wildfire and post-fire erosion events (Shakesby, 2011). For example, the Mediterranean Basin is poorly represented in this meta-analysis, with the exception of the western part of


**Fig. 5.** Photographs of common ground cover and barrier treatments included in the meta-analysis: a) straw bale transport by helicopter (2005 School Fire, Washington, USA); b) helimulching dropping (2015, Navia de Suarna, Lugo, Spain); c) aerial hydromulch application (2002 Hayman Fire, Colorado, USA); d) wood shred mulching ground cover frame (2012 Waldo Canyon Fire, Colorado, USA); e) wood strand mulch study plot (2002 Hayman Fire, Colorado, USA); f) wood chips mulch (2006, Soutelo de Montes, Pontevedra, Spain); g) bark strand mulch (2010, Fial das Corzas, Pontevedra, Spain); h) straw bale check dams in an ephemeral channel (2010 Twitchell Canyon Fire, Utah, USA); i) log barriers (2002 Canyon Fire, California, USA); j) straw wattle barrier (2000 Bitterroot Complex Fire, Montana, USA); k) barriers made with shrubs (2006, Soutelo de Montes, Pontevedra, Spain); l) log dam (2015, Cinco Villas, Orés, Spain).

the Iberian Peninsula. Furthermore, no post-fire erosion mitigation studies were encountered from Mediterranean-type climate regions in Australia, South Africa or South America.

In terms of scale, it is evident that most of the field experiments involved plot scale studies, with few observations from catchment scale studies. Such a distribution is perhaps not surprising due to the increasing complexity, logistic challenges and costs of field erosion studies at increasing scales (Shakesby and Doerr, 2006). The very limited number of studies on post-fire runoff and erosion mitigation on the swale and catchment scale did not allow to meta-analyze the role of spatial scale in a sensible manner, i.e. across the full range of scales from micro-plots to catchments. Hopefully, this study can be used to justify the urgent need for more swale- and catchment-scale mitigation studies, especially since state-of-the-art post-fire emergency stabilization management as operationalized in the USA and Galicia typically targets the risks of the potential off-site impacts of post-fire runoff and erosion. Likewise, this study provides ample justification for more studies on post-fire runoff mitigation across all spatial scales, including the micro-plot and hillslope scales (continuous post-fire runoff and sediment records hardly exist, except from rainfall simulation experiments), especially also since downstream flooding risk is often a major concern in operational emergency stabilization management. The comparative lack of post-fire runoff as opposed to erosion data was also referred in earlier post-fire review studies (Shakesby and Doerr, 2006; Vieira et al., 2015).

The methodological approaches taken by researchers when presenting their results also led to variability in the database. For example, the rainfall data comparable in the majority of the studies corresponded to annual amounts (e.g. Fernández and Vega, 2016a; Keizer et al., 2018; Prats et al., 2019), while in others to actual rainfall for given erosion events (e.g. Robichaud et al., 2008a, 2008b), or to monitoring periods shorter than one year (e.g. Díaz-Raviña et al., 2012; Hosseini et al., 2017; Lucas-Borja et al., 2019). Another example is the start date following fire on which the different studies started the monitoring. Most of the studies initiated the monitoring when the mitigation measures were applied, but in a smaller number, it was started shortly after the fire and before treatment application (e.g. Fernández et al., 2007; Prats et al., 2012, 2016b). Another time-related variable issue was the elapsed period between fire and treatment application, such that in most of the studies treatments were applied within the first post-fire year, while in some exceptions it extended until the second year (Badía et al., 2015), or even the fourth year (Kim et al., 2008). On the other hand, while most of the studies monitored erosion during one or two years, some others monitored it up to five (Prats et al., 2016a) and even seven years (Robichaud et al., 2013b, 2013c). Consequently, individual studies

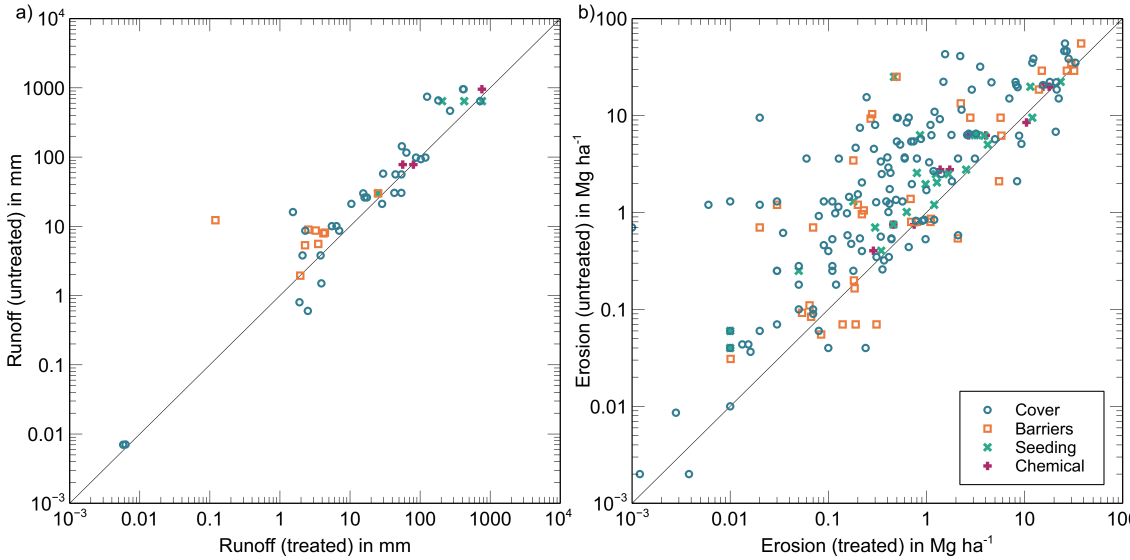


**Fig. 6.** Post-fire runoff (a) and erosion (b) of the respectively 53 and 222 paired observations with and without the various main types of erosion mitigation treatments. The observations are represented on a log-10 scale so '0' values are not displayed.

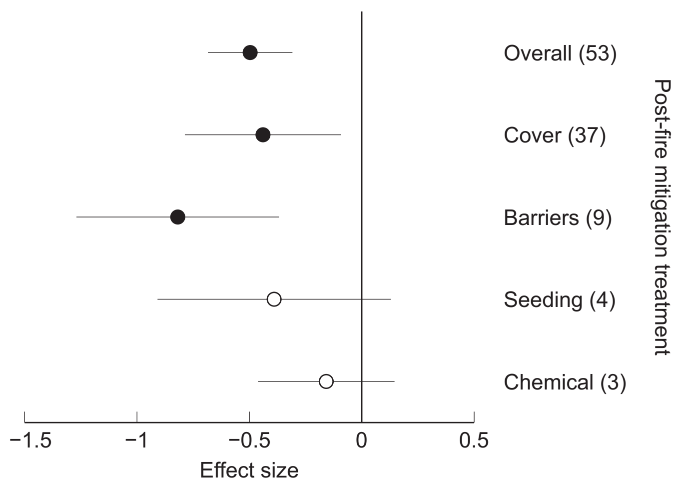


**Fig. 7.** Mean effect size and 95% confidence intervals of all mitigation treatments together ('overall') on runoff as well as of the four main types of treatments, with the number of paired observations given in between parentheses. White markers indicate that the mean effect size was not statistically significant at $\alpha = 0.05$.

might reflect different windows of disturbance periods (Prosser and Williams, 1998; Shakesby and Doerr, 2006), which likely affected the individual effectiveness of the treatments. However, these differences had limited or no implications because both the treated and untreated observations corresponded to the same rainfall conditions and monitoring period.

Regarding the chosen mitigation treatments, there is a strong bias towards the use of cover treatments, which can be explained by the shared agreement among the scientific community about such treatments being the most efficient in preventing post-fire soil erosion (Robichaud et al., 2000; Vega et al., 2013a). However, this limits the comparability of these treatments, since various treatments such as PAM (Inbar et al., 2015; Prats et al., 2014a) or fertilizers (Robichaud et al., 2006), or combinations of treatments such as seeding with mulching (Badía and Martí, 2000; Hosseini et al., 2017; Vega et al., 2015) or seeding with fertilizing (Robichaud et al., 2006) were underrepresented in the database. Such underrepresentation led to the impossibility of calculating an individual effect size with significance for these particular cases or to perform a more detailed comparison across mitigation treatment choices, which would have improved the results of the present study. Another limitation found was that only two of the publications provided the costs of the different treatments, which combined with the uncertainties involved in the estimation of operational costs for different regions, did not allow performing a thorough cost-effectiveness analysis of the studies contained in the database.

### 4.2. *Treatment effectiveness in reducing post-fire runoff and erosion and influence of key-variables*

#### 4.2.1. *Effectiveness of cover treatments*

Previous publications and reports on post-fire management have pointed out the higher effectiveness of cover treatments at reducing soil erosion, especially of straw and wood mulches (Ferreira et al., 2015; Robichaud et al., 2000; Vega et al., 2013a), which is supported by the statistical evidence provided in this meta-analysis. The rationale behind the effectiveness of this group of treatments is the high ground cover provided in the short-term by the mulches, 60–80% being the optimal range for reducing hillslope erosion (Robichaud et al., 2000). However, most of the studies included in the meta-analysis indicate the treatment application rate rather than the initial ground cover it provided.

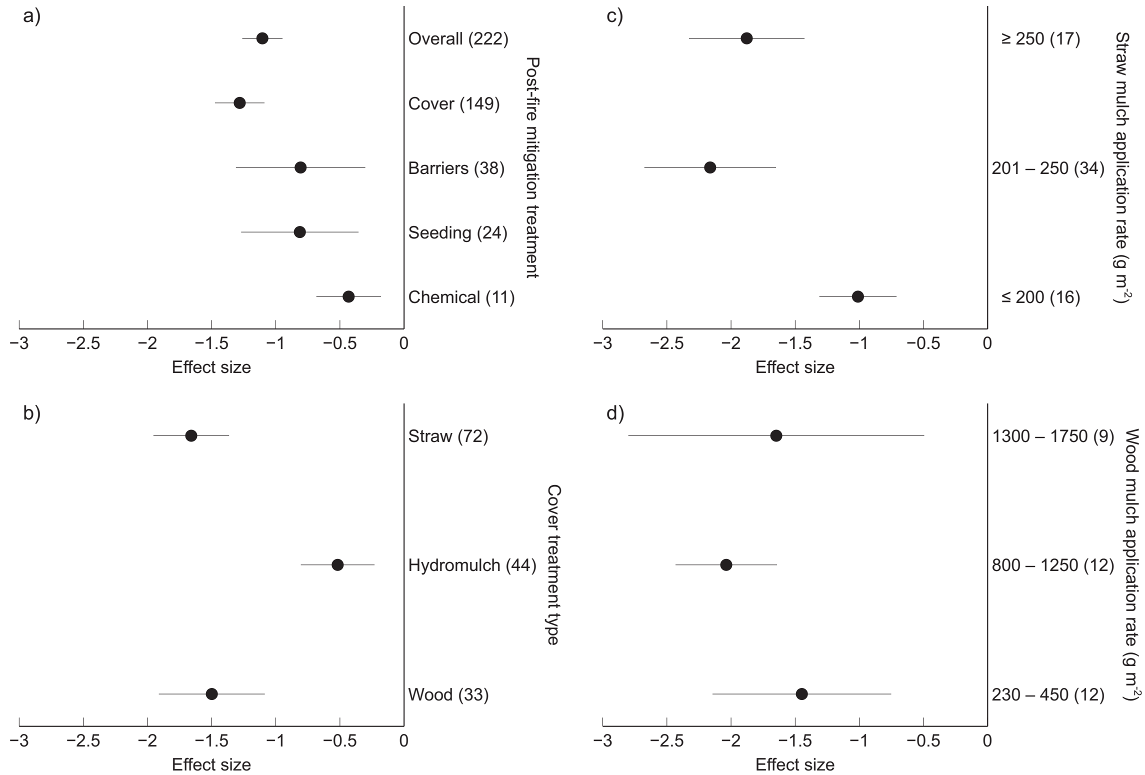


**Fig. 8.** Mean effect size and 95% confidence intervals of a) all erosion mitigation treatments together ('overall') as well as of the four main types of treatments; b) each erosion mitigation cover treatment; c) the different rates of application of straw mulch used across studies: d) the different rates of application of wood materials mulch used across studies. The number of paired observations included in each category is given in between parentheses.

Nonetheless, the results show that straw mulch has a lower efficiency when applied at rates lower than 200 g $m^{-2}$. On the other hand, wood mulches present similar efficiencies among the different application rates, but rates over 1300 g $m^{-2}$ induce more variability in their effectiveness.

Apart from the cover provided by the treatments, another determining factor in their effectiveness is the capacity of the materials to remain on-site during wind and rain events. In this respect, straw mulches are easily decomposed and prone to removal by wind, and its effectiveness is lower with increasing slope steepness (Vega et al., 2013a). For example, Badía and Martí (2000) and Fernández et al. (2011) have reported monthly reduction rates of 4 and 5% in the cover provided by the straw mulch, while De la Fuente and Blond (2010) observed an efficiency reduction of straw mulch in areas hit by strong winds. In the case of helimulching, if the straw is not sufficiently broken up and evenly distributed during application, it can also form clumps, promoting erosion in between them and also limiting the development of vegetation (deWolfe et al., 2008). On the other hand, wood-residue mulches are more resistant to the aforementioned factors and the heterogeneity in size provides diverse protection mechanisms, as the materials absorb rainfall impact and also trap and reduce the movement of sediments (Faucette et al., 2007). Hydromulches are also a good option for treating short steep and exposed slopes because they bind with the soil, being more resistant to wind and water, but are known to decompose quickly, usually within the first post-fire year thus reducing their effectiveness, and their application is rather expensive (Hubbert et al., 2012; Prats et al., 2016b; Robichaud et al., 2013b). All things considered, the results of the present meta-analysis suggest significantly higher efficiencies of straw and wood mulches compared to hydromulch.

### 4.2.2. *Effectiveness of barriers*

There is a general agreement on the effectiveness of barrier treatments in reducing runoff, although it has been scarcely documented (Robichaud et al., 2000) and only two of the publications included in the present meta-analysis dealt with this matter (Badía et al., 2015; Kim et al., 2008), showing opposed results. The contrasting and thus inconclusive results on barrier efficiencies in reducing runoff could be related to the rainfall regimes, observing that they were more effective in a Mediterranean semi-arid area with an average annual rainfall of 346 mm (Badía et al., 2015), than in a more humid area with 2167 mm where barriers had no effect (Kim et al., 2008). It has also been previously indicated that barriers might be more effective for low to moderate rainfall intensity events (Robichaud et al., 2005), as observed in Badía et al. (2015) where the efficiency of barriers decreased with increasing rainfall intensity. However, the differences could also be related to certain issues usually attributed to the implementation of this measure, as the type of materials used, their disposition, and the degree of ground contact (Badía et al., 2015; Wagenbrenner et al., 2006). In the present meta-analysis, barriers were shown to be relatively more effective at reducing runoff than at preventing soil erosion, and also showing certain variability of results. This variability could be related to a matter of design, as the effectiveness of this type of treatments heavily depends on the density of piled materials and thus, the percentage of total sediment delivery they are able to retain, as well as the total storage capacity, which will determine its lifespan (Robichaud et al., 2000).

### 4.2.3. *Effectiveness of seeding*

As opposed to the other groups of treatments, seeding does not provide an immediate protective effect and for this reason, despite it being often used as post-fire erosion mitigation treatment, it is identified as a rather ineffective measure, especially in the first year. However, the full potential of this measure is still uncertain because of the reduced number of investigations studying seeding effectiveness beyond the second post-fire year, as some studies have shown that it reaches its peak of effectiveness from the fifth year on (Peppin et al., 2010 and refs. therein). Seeding treatments have proved to be more effective in areas

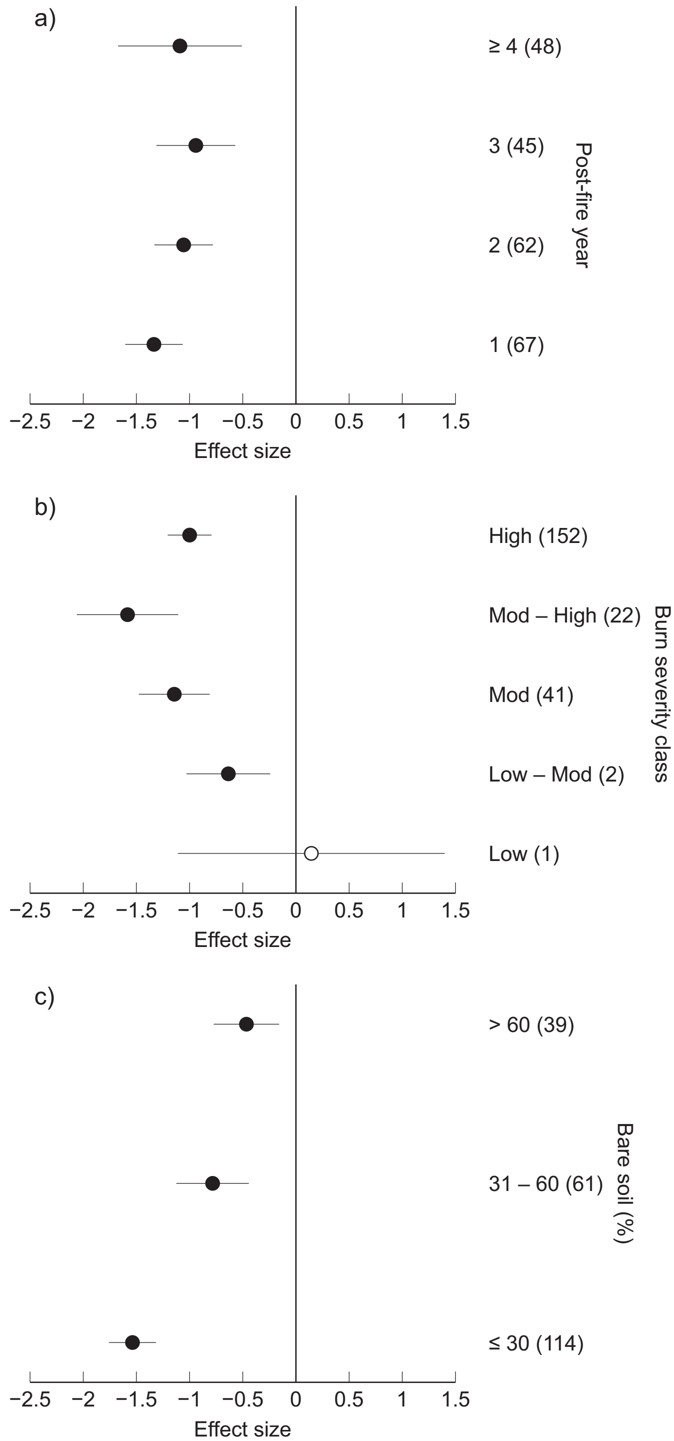


**Fig. 9.** Mean effect size and 95% confidence intervals of post-fire soil erosion mitigation treatments according to: a) the post-fire year; b) burn severity and c) ground cover. The number of paired observations included in each category is given in between parentheses. White marker indicates that the mean effect size was not statistically significant at $\alpha = 0.05$.

with frequent episodes of light rain as opposed to areas in which extreme rainfall events remove the seeds and/or there is water repellency, thus limiting the germination (Fernández et al., 2012; Vega et al., 2015). The present results as well as prior studies indicate that seeding is more effective when combined with other treatments (Badía and Martí, 2000; Prats et al., 2016b). This is especially true for other treatments that immediately provide a protective soil cover and, thereby, not only avoid soil (fertility) losses but also the transport, accumulation or export of the seeds (Robichaud et al., 2006), ensuring the long-term effectiveness of seeding (Peppin et al., 2010).

#### 4.2.4. *Effectiveness of chemical treatments*

The studied chemical treatments, namely the application of polyacrylamides (PAM; Inbar et al., 2015; Prats et al., 2014a) and fertilizers (Robichaud et al., 2006) were the least represented across the data base and showed the lowest effect in reducing erosion and no significant effect on runoff.

The outcome of the PAM effect analysis was dependent on the only

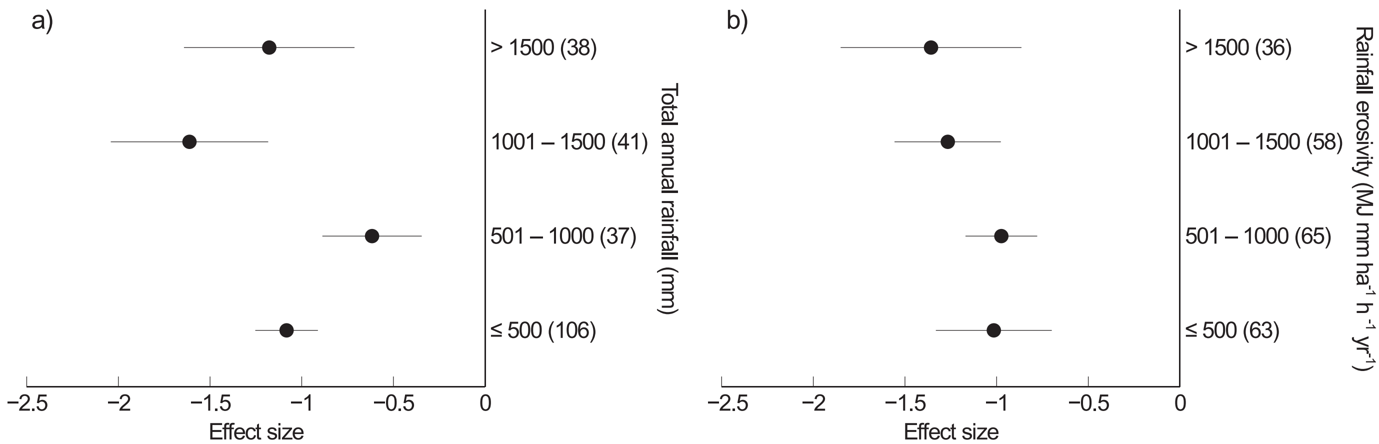


**Fig. 10.** Mean effect size and 95% confidence intervals of post-fire soil erosion mitigation treatments according to: a) total annual rainfall and b) rainfall erosivity. The number of paired observations included in each category is given in between parentheses.

two publications in the database covering this treatment, so it is difficult to draw strong conclusions. In the study by Inbar et al. (2015), PAM had no effect on runoff production but reduced erosion by 35–57% compared to the untreated plots, which showed the expected positive effect in soil structure stabilization. On the other hand, Prats et al. (2014a) reported that PAM reduced neither runoff nor erosion except for a few rainfall events, and attributed this effect to the polymer binding with ash instead of soil particles. Regardless the results, all of the authors agreed on the need for PAM to be wetted when applied to maximize its effect, favoring its incorporation into the soil and strengthening the adsorption during the dry cycles.

The application of fertilizers neither alone nor in combination with seeding had any effect on reducing the post-fire erosion rates, which was justified by not attaining an effective ground cover, especially during the first post-fire year (Robichaud et al., 2006). In that same publication, the authors claimed that the effectiveness of that treatment or combination of treatments heavily relies on the occurrence of light rainfall events that promote the development of vegetation cover shortly after fire, which would then protect against erosion events in the long-term.

#### 4.2.5. Influence of key-variables in treatment efficiency

Besides the technique used to mitigate the increased post-fire runoff and erosion, there are several other factors that might influence treatment efficiency. Such factors can be considered when choosing the best treatment for a given place, time or burn severity characteristics.

Although it is generally accepted that the sooner treatments are applied the better, the meta-analysis results did not indicate a clear difference in mitigation efficiency between the years of their application. However, the importance of treating burned areas in an early stage was identified by several researchers as fundamental to ensure minimal soil losses during the post-fire period (Fernández et al., 2016a; Robichaud et al., 2013b). In addition, the mitigation measures should also ensure erosion reduction in the post-fire years, as revealed by several studies with lasting windows of disturbance, especially in high burn severity areas (Badía et al., 2015; Prats et al., 2016a, 2016b; Robichaud et al., 2013b).

It is now also well established that burn severity is important in the post-fire hydrological and erosive response (Vieira et al., 2015). Areas burnt at high severity are typically the focus of priority in operational emergency stabilization management in the USA and Galicia. The observed trend in the overall efficiency of post-fire mitigation treatments showed a concomitant increase with burn severity from low-moderate to moderate-high classes. However, when the burn severity exceeds the high severity threshold, there is an apparent reduction in the effect of the treatment, highlighting that the efficiency peak of the mitigation treatments does not match the highest possible impact (Fig. 8b).

The presence or lack of a protective ground cover has been frequently identified as a key-variable in hydrological and erosive processes (Prats et al., 2016b; Vieira et al., 2018). In our case, the chosen variable was bare soil surface, and its combination with the mitigation treatment presented one of the clearest messages of this study. Regardless of the treatment, the bare soil surface was significantly linked to mitigation efficiency, and ensuring a bare soil cover below the 30% threshold is fundamental to obtain the maximum mitigation. This observation agrees with the previous recommendations by Prats et al. (2014a), Robichaud et al. (2000), Ferreira et al. (2015) and Vega et al. (2013a), for the implementation of post-fire mitigation measures.

Rainfall often plays a key role in post-fire runoff and erosion processes (Robichaud et al., 2013b; Prats et al., 2012). However, annual rainfall amount was not found here to affect mitigation efficiency in a clear manner. This could reflect the distinct climate regions of the observations, with similar rainfall totals masking differences in rainfall intensity. Rainfall intensity can be a more important driver than rainfall totals in post-fire hydrological and erosive processes, especially over short monitoring periods (Malvar et al., 2017; Robichaud et al., 2013b; Vieira et al., 2018). Unfortunately, our database did not allow retrieving a representative indicator for rainfall intensity, because it included studies with variable field monitoring periods and time-steps. Therefore, the present meta-analysis used an existing rainfall erosivity dataset as an alternative indicator or rainfall intensity, finding the effect size of post-fire erosion mitigation treatments to increase with rainfall erosivity. We therefore strongly recommend that future post-fire erosion mitigation studies explicitly include information on rainfall intensity or erosivity, even when rainfall total is the main runoff-erosion driver in their specific cases.

### 4.3. Management implications and recommendations

Increasingly, society demands actions to mitigate the risk of post-fire erosion and runoff, fundamentally for maintaining water quality and protecting populations and other values at risk. Despite the successful progress made in some regions in the application of treatments, there is still a need to disseminate results so that land managers do not use techniques that are relatively ineffective at mitigating risk or have a lower cost-effectiveness. The importance of fire severity in the selection of areas for erosion mitigation treatments must be better conveyed to forest and land managers to establish a clearer consensus regarding where and when which treatments will provide the most benefit. This is particularly true for areas burned at low or moderate severity, where adequate ground cover is provided by remaining forest floor material and natural mulch such as scorched coniferous needles. These factors should be duly considered, because of the considerable costs of post-fire treatments and the typical impossibility to apply treatments everywhere and before the occurrence of significant post-fire rainfall.

Techniques that provide a protective cover and attempt to emulate the ground cover that has been consumed by fire are the most effective in reducing soil erosion. In this sense, agricultural straw seems to be hard to replace, because high amounts can be obtained locally and with relative ease. Other materials, produced locally or not, are more difficult to obtain and spread throughout burned areas. In situ mastication of non-commercial juvenile trees is promising but may be insufficient to protect the soil if tree density is low (Fernández et al., 2019a). The use of agricultural straw can produce the risk of introducing non-native plants, but evidence shows that it is low in communities dominated by resprouters (Fernández et al., 2019b) and the mulch cover may be beneficial for the recovery of the vegetative cover in places with water stress during dry summer seasons (Fernández et al., 2016b). Using local material could be a feasible solution, although it would demand planning the needs in advance. Despite the existing evidence that barriers are less effective in reducing post-fire erosion under intensive rainfall, the construction of barriers remains a common hillslope treatment. Alternatively, barriers may continue to be the preferred solution where and when there exist legal obligations or practical needs to manage the standing biomass of burned trees and shrubs or logging residues. Arguably, barriers could be most useful for reducing runoff velocity where runoff becomes concentrated (e.g. due to the road drainage network), especially when combined with mulching on the hillslopes with elevated runoff-erosion risk.

### *4.4. Implications for future research*

This review presents a cumulative assessment of common post-fire treatments. Although the studied geographic regions encompass a small portion of the fire-prone areas globally, they represent an area of greatest concern due to human population and consequences of major flooding, erosion and debris flow events post-wildfire. This review highlights the use of mulches as an effective post-fire erosion mitigation treatment, especially agricultural straw mulches, but also different types of wood-based mulches. These wood-based treatments are being implemented on steeper slopes (>60%), burned at high severity, and have the potential to resist high winds, so they can be used at high-value sites. Nevertheless, other emerging treatments or combinations of treatments would benefit from further research to compare with traditional measures.

Combinations of treatments have received much less rigorous evaluation than single-treatment application. deWolfe et al. (2008) combined hillslope mulching with channel treatments involving log erosion barriers, high rates of straw mulch, seeding, check dams and debris racks after the 2002 Missionary Ridge Fire in Colorado. deWolfe et al. (2008) reported little sediment reaching their reservoir, which was the public drinking water supply for the city of Durango, Colorado. Another beneficial addition would be combining agricultural straw mulch with a secondary treatment of a tackifier to hold the straw in place, which has been studied in a wind tunnel experiment but not in field trials (Robichaud et al., 2017). Currently, other novel treatments, namely the use of geotextile tubular containers filled with organic residues and/or (techno-)soil (geotubes), are undergoing their first field trials in recently burnt areas from North-Central Portugal and NW Spain. Serpa et al. (2020) referred the testing of geotubes that are envisaged to act as a barrier against soil erosion, as well as to enhance post-fire vegetation recovery through seeds and mycotechnosoil (mycorrhized technosoils) contained in the geotubes, together with straw. Nevertheless, additional studies are needed to justify the extra expense of combining treatments as this is often cited as a reason not carry out this more complex approach.

Alternative chemical treatments for erosion control, such as polyacrylamide, guar gum and xanthan gum are being researched as the chemical composition can be altered to meet specific soil types by bonding better to the soil matrix. Polyacrylamides are synthetic binders from the petroleum industries, guar gum is extracted from legumes, and xanthan gum is a polysaccharide that is produced by bacterial fermentation. With any chemical treatment, short- and long-term effects on aquatic and terrestrial ecosystems need to be evaluated.

Biochar is being implemented on very disturbed forest sites (i.e. mine reclamation) yet it might have a role in post-fire environment. This product can be obtained from burned trees such as trees cut for roadside hazard reduction or nearby forest thinning projects. Several laboratory rainfall simulation studies have shown soil erosion reduction following biochar application to unburned soils (Abrol et al., 2016; Sadeghi et al., 2021). Very recently, a first field study tested the combined effectiveness of biochar application with straw mulching to reduce post-fire erosion (Prats et al., 2021).

Recently, wildfires have burned through riparian areas with flames reaching the stream channel (Robichaud et al., 2020a). Treating these riparian areas has not received much attention in the literature as historically they have not burned severely or at all due to high soil moistures near the channels. However, treating riparian areas could have specific unwanted outcomes such as effects on stream chemistry or vegetation regrowth.

Since wildfires do not only affect forests, it is needed to address reducing erosion, runoff and contaminate transport from urban areas in the post-fire environment. Household chemicals, plastics, and heavy metals can all become mobilized in urban runoff and have significant effects on storm water quality. However, the evaluation of treatment effectiveness on water quality parameters is lacking in the literature.

There are only a few studies that address longer-term effectiveness of treatments as it is difficult to receive funding for long-term projects. Robichaud et al. (2020b) suggested that wood mulches can last 10 years or more and were found to be still effective after this time at the 2002 Hayman Fire in Colorado. Bontrager et al. (2019) studied agricultural straw mulch up to 10 years after fire and found little effect on tree seedling establishments, while Jonas et al. (2019) found some effect of mulch on tree seedling establishment and little longer-term effect on nitrogen availability. Additional regional studies are needed to ensure that short-term erosion control is not causing a long-term ecological problem.

One of the limitations found when conducting this meta-analysis was the underrepresentation of swale and catchment-scale studies. It is necessary to conduct more studies on the effectiveness of post-fire soil erosion mitigation treatments at catchment scale, which is the scale with higher interest from the post-fire management perspective (Robichaud et al., 2013c), including the monitoring of runoff, surface erosion, channel processes, and debris flows. Doing so would also allow a better understanding of the erosive response at catchment scale on a rainfall event basis after the application of mitigation treatments. In addition, it would increase the available data to compare the effectiveness of mitigation treatments across scales.

The studies included in the meta-analysis generally lacked information about the application costs of the different mitigation treatments. Such information, however, would be most useful as the treatments' cost-effectiveness could be a key element in operational emergency stabilization management to decide which measures to apply where and with which priority. Therefore, we propose that future studies include such information, even if just in the supplementary materials.

### *4.5. Modeling as a tool for optimizing the implementation of post-fire mitigation treatments*

Computer models allow for predicting the effects of treatments (Robichaud et al., 2016), yet more validation studies are needed to build confidence among the post-fire assessment communities on the efficacy of treatments reducing threat of flooding, erosion, degraded water quality and downstream sedimentation. A recent review has highlighted that the implementation of post-fire mitigation measures has been scarcely addressed in post-fire soil erosion modeling studies (Lopes et al., 2021). However, further investment in such methodology could potentially leverage the use of mitigation measures for post-fire management planning or for research purposes. From a management perspective, modeling could be used for treatment implementation optimization in a recently burned area, since hydrological modeling can integrate various sources of variability (e.g. climate, burn severity, topography) in a single prediction, or even produce scenarios considering other important aspects such as available funds or technical means. From a research perspective, hydrological modeling could also advance in the understanding of how specific mitigation techniques affect hydrological processes thus eliminating sources of uncertainties from field experiments, and possibility finding other solutions or mitigation configurations based on simulations before their testing in the field. Having that said, we believe it is important to direct efforts towards integrating mitigation measures in post-fire modeling applications, and for the widest range of mitigation techniques possible.

## 5. Conclusions

The present systematic review of the scientific literature on post-fire runoff and soil erosion mitigation treatments is the first review that includes a quantitative assessment, using meta-analysis, of the effectiveness of these treatments across the full body of field studies in all their diversity and variety. The principal findings of the meta-analysis with respect to the effect size of the tested treatments are as follows:

- cover (of straw, wood and hydro-mulch) and barrier treatments significantly reduce post-fire runoff;
- all tested mitigation treatments significantly reduce post-fire erosion but cover treatments are the most effective, followed by barriers and seeding while chemical treatments are the least effective;
- straw and wood mulches are equally effective at reducing post-fire erosion but more effective than hydro-mulch;
- straw mulch is less effective at reducing post-fire erosion at application rates below 200 g $m^{-2}$ than between 200 and 560 g $m^{-2}$;
- wood mulch is similarly effective at reducing post-fire erosion over a wide range application rates, from 230 to 1750 g $m^{-2}$, but the effect size tends to be more variable and, hence, more uncertain at application rates above 1300 g $m^{-2}$.

In addition, the meta-analysis results allowed drawing the following main conclusions with respect to the role of selected key variables in the effect size of the post-fire erosion mitigation treatments:

- time-since-fire: the treatments' effect size does not vary significantly with the post-fire year of their implementation;
- burn severity: the treatments' effect size tends to increase from low to moderate burn severity but to decrease again from moderate to high severity;
- ground cover: the treatments' effect size varies significantly with the protective ground cover that is achieved, with the effect size being at its maximum when bare soil cover is below the 30%;
- rainfall: the treatments' effect size does not vary significantly with annual rainfall amounts but there exists a trend towards greater effect size increasing rainfall erosivity.

It can be concluded that further efforts are needed on conducting field studies, especially at larger scales and in a higher variety of geographical locations, and validate model results with field exercises, analyzing new emerging treatments and different combinations of measures to ensure the optimal actions are taken after wildfires.

## Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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