Tel-Aviv University

Porter school of Environmental studies

Using synthetic polymers for soil erosion control after forest fires

Thesis submitted towards the M. Sc. Degree

By

Assaf Inbar

The research was performed in the Department of Molecular Biology & Ecology of Plants at Tel-Aviv University, and in the Department of Soil, Water and the Environment, the Volcani Center.

Under the supervision of

Prof. Marcelo Sternberg

Dr. Marcos Lado

Dr. Meni Ben-Hur

August 2011

This work is dedicated with love to my father, Mr. Ya'acov Inbar, who inspired me to follow my dream to be a scientist, but did not survive to see it completely fulfilled.

Thank you father for helping me to be who I am today. I owe this all to you.

Table of Contents

1   Abstract

Forest fires are a major concern, especially in the Mediterranean region where climatic conditions of long hot summers and mild winters favor outbreak of wildfires, and especially in a period of worldwide climate change, and changes in land use. Forest fires make changes to the environment by the reduction of vegetation cover and altering properties of the forest soils, which together, can make changes to the hydrological cycle. Forest fires are different from one another in terms of fire severity. Moreover, different fire severities often occur within a certain fire, which depend of different climatic conditions, topography and spatial distribution of combustible fuel load. The aforementioned situation makes mosaic patches of fire severity that differently affect the underlying soil. One of the major effects of wildfires is the increase in runoff and consequently soil erosion before vegetation recovery. Anionic polyacrylamide (PAM)

is a soil amendment that had been proven effective in reducing runoff and soil loss in arable lands and in stabilizing steep slopes. PAM is cheap and is applied in relatively small quantities to achieve positive effect. Only a few studies have tested PAM as a post-fire erosion control method. Moreover, the effect of PAM on stabilizing soils of different fire severity had never been tested. The objectives of this study were (i) to test the effect of different fire severities on pysicochemical properties of Pale rendzina soil, and their influence on infiltration rate, runoff and soil loss interactions; and (ii) test the effect of anionic polyacrylamide (PAM) as a post-fire erosion control method on the previously studied soil from different fire severity, in laboratory controlled experiments and in preliminary field experiments in a burnt area; and (iii) test the effect of two rates of PAM application on vegetation recovery using wheat (*Triticum aestivum* L.) as a control plant for herbaceous annual vegetation recovery.

In order to test the effects of different fire severities we used Pale rendzina forest soil from Birya forest that was subjected to different fire treatments and represent some of the heterogeneous fire conditions that may exist after a wildfire, and their effect on the underlying soil. Unburnt treatment (UB soil), represents a soil from a location that the fire did not have any effect on the underlying soil; direct fire treatment (DF soil) represents a soil from a location that was directly affected by a low-moderate severity fire; and heat treatment (HT soil) represents a soil from a location that was affected by prolonged heating conditions in moderate temperature but with no direct contact with the flames. Pale rendzina soils were sampled from outside and inside (UB and DF soils, respectively) an area of low-medium fire severity burnt forest in Birya, Israel. Half of the unburnt soil was burnt in a muffle at $300^{o}c$ for 8 hours to create the HT soil. The HT soil was found to make changes in the soil physicochemical properties of the soil compared to the UB and DF soils. Among the changes that were found was a decrease in organic matter (OM) content, sodium adsorption ration (SAR), Cation exchange capacity (CEC) and in clay fraction, and an increase in carbonate content ($CaCO_3$), in the dissolved organic carbon (DOC) and the electrical conductivity (EC) of the soil solution. The DF soil did not differ much from the UB soil, except in having higher OM content and CEC, and lower SAR in the soil solution.

Three storm-rainfall simulator experiments of 80mm in 47mm h^{-1}, with a 72 hr drying period in between, were conducted with the different soil samples. Differences in infiltration rate (IR), runoff and soil loss were found between the three fire treatments. Very low water

repellency (WR) was found in all soils, rejecting it as a possible cause for the differences. Even though its OM content was decreased, the HT soil had shown the highest IR, lower runoff and soil loss amounts in the 1st rainstorm. These results were suggested to be a result of higher structural stability due to fusion of clay particles in the aggregates, and due to physicochemical changes in the soil solution that resisted the dispersion of clay. The fusion of clay particles resulted in stronger aggregates and was proven by a slaking test. Furthermore, as found by a dispersion test, clay dispersion was reduced due to higher EC and lower SAR combined with the physical stabilization, and resulted in stronger structural stability. Although the DF soil had lower IR than the HT soil, it showed higher IR and lower runoff and soil loss than the UB soil in the 1st rainstorm. Nevertheless, no significant differences were found between DF and UB in the slaking and dispersion tests. However, dispersion of the DF soil was slightly lower, probably caused by lower SAR values, and might be the source of difference in IR/runoff/soil loss between the two soils. The lack of difference in the stability tests also explains that the higher OM content of the DF soil as compared to the UB soil does not take part in stabilizing aggregates. The increase in OM content is most probably a result of exterior input of OM that was not combusted completely, and was mixed with the soil in the sampling process. Similar to the HT soil, the differences between the UB and the DF soils in IR diminished in the 2nd and 3rd rainstorms but reached similar values in the 3rd rainstorm. We concluded that in Pale rendzina soil, the exposure to different fire conditions stabilize the soil structure by changes in physical and physicochemical properties of the soil and soil solution, thus increasing their resistance to the destructive forces of the raindrop impact, fast wetting, clay dispersion and consequently seal formation. Nevertheless, the differences were reduced in consecutive rainstorms indicating that the stabilization does not last long, especially when the soil remains bare as a consequence of the destruction of the vegetation cover. Thus, we suggest that soil erosion control methods should be implemented.

Similar rainfall simulator experiments on the same fire treated soils with addition of 50 kg ha^{-1} granular PAM application were conducted. Testing the PAM on the DF and UB soils, it was found in the 1st rainfall event after the application of PAM, the IR significantly decreases compared to corresponding control treatments, causing a significant increase in runoff due to its high viscosity. Nevertheless, a reduction in soil loss was observed. In consecutive rainstorms, the PAM treatments yielded less soil loss than the control while IR increased and runoff decreased.

We suggest a two stage mechanism that enables the reduction in soil erosion with application of PAM: (i) the increased viscosity of the runoff in the 1st rainstorm cause it to be less erosive due to a decreased velocity and transport capacity, thus detachment is reduced, and less soil is eroded with the runoff. Furthermore, the flocculation effect of the PAM molecules flocculate fine particles carried by the runoff, further reducing the transport capacity; (ii) after the drying period, PAM molecules that were not washed away by the runoff adsorb to soil particles and stabilize them, causing the increase in IR and reduction in runoff and soil loss. The latter stabilization effect was proven by slaking and dispersion tests. Unlike the two previously mentioned soils, the HT soil showed a different pattern, as no reduction in soil loss in the 1st rainstorm was found. The cause for the aforementioned finding was assumed to be a lower viscosity of the runoff caused by higher EC values compared to the other two PAM-treated soils, which might have caused an increase in its erosive forces and increase soil loss on one hand, but the runoff was still viscous enough to reduce infiltration and increase the runoff volume. Nevertheless, in consecutive rainstorms, IR was increased significantly, resulting in lower runoff and a reduction in soil loss. These results indicate that the second mechanism was effective in reducing soil erosion despite the high stability of this soil. Testing the same PAM treatment on a soil with different texture and mineralogy (a non calcareous kaolinitic sandy loam soil, from Barbanza Galicia, Spain) that was treated with the same treatment as the HT soil showed that both mechanisms were active in reducing soil erosion.

In order to test the effect of PAM under field conditions, twelve runoff plots were built in the previously mentioned burnt area, on a westerly slope with ~40% descent. Three treatments with 4 replicates each were tested: untreated control and; 25- and 50 kg ha^{-1} granular PAM application. Ash deposits on the soil surface were carefully removed in order to test the effect of PAM on the mineral soil alone. The PAM application took place close to the beginning of the winter of 2009-2010 by manually spreading it on the soil surface. After almost every rainfall event, eroded sediments that were collected in funnels located at the bottom of every plot were collected, runoff volume was measured and runoff samples were taken to determine the sediment concentration. Rain depth of every rain event was recorded using five rainfall gauges. A linear correlation was found between cumulative rainfall and cumulative runoff. Although not statistically significant, the 50 kg ha^{-1} treatment was found to be the most effective in reducing runoff and soil loss, reducing soil loss by 50% compared to untreated control. The 25 kg ha^{-1}

treatment reduced soil loss by 23%, but with slightly higher runoff amounts compared to control. The viscosity and flocculation mechanisms that were identified in the rainfall simulator experiments were found to be active during the 1st rainstorm, but the stabilization mechanism was more effective, resulting in less eroded soil compared to control throughout the rest of the season. Fifteen 0.25 m^2 square plots were built at the same time and near the runoff plots for a vegetation recovery experiment. Thirty seeds of wheat (*Triticum aestivum* L) were seeded in the center of each plot, and PAM was applied in five replicates for each PAM application rates as previously described. All three treatments yielded similar aboveground biomass and seeds production. Thus, we postulated that PAM at the two rates tested, had no effect on aboveground biomass and seed production, and it can be assumed that PAM application at these rates will not affect vegetation recovery, at least not of herbaceous annual grasses.

2   Introduction

In the countries of the Mediterranean basin, fire is one of the main causes of forest destruction. About 50,000 fires burn an area of 700,000 - 1,000,000 ha of Mediterranean forest annually, causing enormous economic and ecological damage (Vélez, 1990). The Mediterranean areas are characterized by long dry summers when the average daytime temperature exceeds 30°C. These conditions may reduce soil water content and forest litter, and as a consequence they favor forest fires. Furthermore, in some regions, inland high speed dry summer winds (e.g. Israeli Sharav) make the atmospheric humidity to drop and may contribute to the spread of fires (Vélez, 2002). Since the 60's, there has been an increase in the number of fires and the total area burnt in the Mediterranean basin (Pausas and Vallejo , 1999), which was attributed mainly to rural depopulation (i.e. combustible fuel buildup after farm abandonment; Pausas and Fernández-Muñoz, 2009, Shakesby, 2011), afforestation with monospecific flammable tree species (Shakesby, 2011), climate change (Pausas et al., 2004, Pausas and Fernández-Muñoz, 2009) and the increase in anthropogenic activities (Giovannini et al., 2001).

In 2010, the total forest area in Israel sums up to 154,000 ha, which represents 7% of the total area of the country. Over the last 100 years, extensive afforestation efforts have been made, and nowadays 57% of these forests are planted (Main report FAO Forestry Paper, 2010). Planted forests cover 8% of the country's Mediterranean area, where annual rainfall amount ranges from 300 to 900 mm year^{-1} in a variety of soil types and topography (Osem et al., 2008). These planted forests consist mainly of three Coniferous species (*Pinus halepensis*, *Pinus brutia*, and *Cupressus sempervirens*) and one eucalypt (*Eucalyptus camaldulensis*) species (Ginsburg, 2000).

Similar to other Mediterranean areas, an increase of fire events and total burnt area has been observed in Israel during recent years (Wittenberg and Malkinson, 2009). The origin of these fires reveals a prominent human component, through both negligence and arson. The main causes of accidental fires in Israel are: hikers who light fires, military units in training, wars (such as the 2006 second Lebanon war) and farming activities. But the most important cause of forest fires in Israel is arson (IFFN No. 15, 1996).

There are several types of forest fires: i) a surface fire burns aboveground fuel load; ii) a ground fire burns the organic material in the upper soil as well; and iii) a crown fire advances through the top of trees or shrubs relatively independent from the surface fire (DeBano et al.,1998). To evaluate the consequences of a fire, two are the main parameters considered: fire

intensity, which is the rate of energy per unit of time and length of the fire front; and fire severity, which describes the ecosystem responses to fire (DeBano et al., 1998; Pausas et al., 2003; Certini, 2005, Keeley, 2009). During a wildfire, local variations in soil properties, lithology, plant composition, fuel load and its moisture content, topography and climatic conditions result in an heterogeneous spatial distribution of fire severity, leaving a mosaic of areas barely affected by the fire alternating with others seriously impacted (Kutiel et al., 1995; Lavee et al., 1995). Pauses et al. (2003) defined three levels of fire severity that are determined by the damage made to the canopy of the trees: (i) in low severity fires, > 20% of the leaves remain green and the tree is still alive; (ii) in moderate severity fires, trees are dead and mainly brown-colored with >80% of the canopy scorched but not consumed (<5% of green leaves may occur on top, and some leaves at the bottom are consumed); and (iii) in high severity fires the trees are dead, >80% of the leaves on the canopy are consumed and the rest are scorched (Pauses et al., 2003).

2.1 Effects of fire on forest soils

Forest fires can affect the entire ecosystem, as a result of the release of a variable amount of thermal energy during the combustion of biomass and organic residues.

During the combustion process, heat is transferred to the surroundings, moving downward, and transferred to the litter and duff layers, or directly to the mineral soil if these layers are not present. If these layers are present, they can be ignited and undergo combustion themselves, which generates additional heat that is transferred downward into the underlying mineral soil (DeBano et al., 1998). The increase in temperature can affect the physical, chemical and biological properties of the forest soils (DeBano et al., 1998; Certini 2005).

The effect of fire on the forest soil depends directly on the severity and duration of the fire. These factors are controlled by several environmental factors that affect the combustion process, such as fuel properties and spatial distribution, air temperature and humidity, wind speed and topography of the site (DeBano et al., 1998). Heat in moist soils is transported faster and deeper in the soil profile, but do not exceed 95°C until the water completely vaporizes (Campbell et al., 1994). Temperatures can rise then to 200-300 $^{\circ}$C, although if heavy fuels loads are present the temperatures can reach 500-700°C at the soil surface (DeBano et al., 1998), but can get as high as 850°C instantaneously (DeBano, 2000). The combination of combustion and heat transfer produces temperature gradients that depend on thickness, packing and moisture of

the fuel bed (Certini, 2005). After fires, soil temperatures can remain high from few hours up to several days (Certini, 2005).

2.1.1 Fire effects on physical and chemical properties of the soil

As mentioned before, wildfires are known to alter physical and physicochemical properties of the soil (DeBano et al., 1998). One of the soil properties affected by the increase of temperature associated to wildfires is soil organic matter (OM). The destruction of soil OM during combustion was found to cause a reduction of the cation exchange capacity (CEC) of the soil (Giovnnini et al., 1990; DeBano et al., 1998). Moreover, the modification of OM can alter soil structure, since this soil property is essential for the formation and stabilization of soil aggregates, particularly in the surface horizons where most of OM is concentrated (Tisdal and Oades, 1982; Chaney and Swift, 1984). Through combustion of OM, a moderate-high severity fire can cause a modification of aggregate stability and soil structure, an increase of bulk density, and a decrease of porosity, which in turn may decrease infiltration and water holding capacity, and may increase runoff and soil erosion (DeBano et al., 1998). The impact of heat on soil aggregates depends on the intensity and duration of exposure to the heat and on the original physical and chemical properties of the soil. For example, aggregate stability decreases dramatically when at high temperatures organic cements within the aggregates are disrupted (Garcia-Corona et al., 2004; Varela et al., 2010), but the remaining aggregates can show higher stability than the original ones due to formation of cementing oxides and fusion of clay minerals (Giovannini *et al*., 1988; Giovannini and Lucchesi, 1997; Badía and Martí, 2003).

Fires can also increase the loss of nutrients by volatilization, mineralization, erosion, runoff and leaching (DeBano et al., 1998). The impact of fire on the nutrients in ecosystems depends greatly on fire severity (Neary et al., 1999). For example, low severity fire can cause short term increases in the availability of nutrients (Kutiel and Shaviv, 1992; Kutiel and Inbar, 1993; Hernandez et al., 1997; Neary et al., 1999). However, moderate and severe fires can result in large fluxes of nutrients leaving the ecosystem as a result of increased microbial processes and amplified erosion and leaching (Neary et al., 1999, Gimeno-Garcia et al., 2000).

Wildfires were found to temporarily increase pH in Mediterranean calcareous soils (Kutiel and Naveh, 1987; Kutiel and Shaviv, 1989, 1992), mainly by the production of ash the release of soluble cations during the combustion of organic matter, and the denaturation of organic acids and loss of hydroxyl groups in high temperatures (Giovannini et al., 1988; Certini,

11

2005; Terefe et al., 2008). However, Kutiel and Inbar (1993) did not find an increase in pH after a wildfire over Pale rendzina soil, and postulated that the pH was not increased due to the fact the fire was of low intensity. Fires can also cause an increase in the electrical conductivity (EC) of the soil due to the increase of soluble inorganic ions that result from the combustion of soil OM, vegetation and litter (Kutiel and Naveh, 1987; Kutiel and Inbar, 1993; Kutiel et al., 1995; Hernandez et al., 1997; Iglesias et al., 1997; Pardini et al., 2004; Terefe et al., 2008; Tessler et al., 2008).

2.1.2 The effect of forest fire on overland flow generation and soil erosion

Overland flow (or runoff) and soil erosion are natural hydrological, geological and ecological processes, with positive and negative effects on ecosystems and humans. On one hand, overland flow provides water and nutrients to streams (Andreassian, 2004; Bruijnzeel, 2004), and on the other hand it can also cause problems such as water loss, soil erosion, floods, pollutants transportation and other environmental hazards (Johnes et al., 1998; Calder, 2007; Van Dijk and Keenan, 2007). Runoff is defined as the part of the precipitation that does not infiltrate into the soil, and it is generated when the rainfall intensity at the soil surface exceeds the infiltration capacity of the soil. Soil erosion is caused by two main processes: (i) detachment of soil particles from the soil surface by raindrop impact and/or sheer forces of the runoff; and (ii) transport of detached particles by the runoff and splash (Baver et al., 1972).

The changes in soil properties as a result of fire can contribute to an increase of runoff and sediment yield from the burnt area (Prosser and Williams, 1998; Robichaud and Brown, 1999; Moody and Martin, 2001; Benavides-Solorio and MacDonald, 2005). Several studies indicate an increase in runoff and soil erosion in the years after a fire event and before the vegetation cover recovers, being the first winter the most sensitive in terms of soil loss (Inbar et al., 1998; Wittenberg and Inbar, 2009). However, null and positive effects of fire on runoff and soil loss yields it have also been reported (Kutiel and Inbar, 1993; Lavee et al., 1995; Kutiel et al., 1995).

Several factors had been mentioned as responsible for the increase of runoff and soil erosion after forest fires: (i) loss of canopy and litter cover (Inbar et al., 1998; Johansen et al., 2001; Larsen et al., 2009); (ii) soil water repellency (WR-Doerr et al., 2009); and (iii) soil sealing by ash and/or sediment particles (Neary et al., 1999; Martin and Moody, 2001; Larsen et al., 2009).

In both burnt and unburnt areas, the surface cover is important to control infiltration, and to reduce runoff and soil loss (Brock and DeBano, 1982; Renard et al., 1997; Inbar et al., 1998; Larsen et al., 2009). Forest canopy and litter prevent runoff formation and soil erosion by several mechanisms: (i) rainfall interception, in which the canopy of the vegetation and the litter layer decrease the intensity of the rain reaching the soil surface and the erosive forces of the raindrops, thereby reducing particle detachment during a rainstorm (Ben Hur et al., 2011); (ii) contributing to OM content in the soil, which maintains a stable structure and high porosity (DeBano et al., 2005); and (iii) increasing soil roughness that traps detached particles, limiting their transportation, and that reduces the velocity of the runoff, decreasing its erosivity and ability to detach soil particles (Lavee et al., 1995; Ben-Hur et al,. 2011).

Burning has been shown to induce or enhance soil WR (Doerr et al., 2009), a phenomenon commonly cited as a primary cause of the observed post-fire increases in peak flows and sediment yields in some soils (DeBano, 1981, 2000; Robichaud, 2000; Shakesby et al., 2000). However, the effect of WR on runoff and soil erosion is scale dependent, and varies in space and time. Furthermore, it has also been cited that in some cases water repellency exist naturally at the top 3 cm of the soil due to water repellant substances that originate by leaching of the organic horizons to the underlying mineral soil (Hoffman et al., 2001; Doerr et al., 2000; Horne and McIntosh 2000, Doerr et al., 2009a,b). Moreover, fire has also been shown to decompose water repellent substances at temperatures higher than 288^0C (DeBano et al., 1988). It has been found that these substances coat soil particles and by that increase aggregate stability (Mataix-Solera and Doerr, 2004; Varela et al., 2010). However, these substances prevent water to infiltrate the soil, consequently increasing runoff and soil erosion (DeBano, 1981, 2000; Robichaud, 2000; Shakesby et al., 2000). Larsen et al. (2009) found that raked unburnt soil showed exactly the same sediment yield as a burnt soil with a comparable surface cover, rainfall erosivity, and soil WR values, which indicates that post-fire sediment yields were primarily due to the loss of surface cover rather than fire-enhanced soil water repellency. Doerr and Moody (2004) stated that the linkage between soil WR and post fire runoff and erosion rates has rarely been demonstrated. Moreover, not all the soils develop WR as a result of fire (Mataix-Solera et al., 2008).

The importance of ash cover on the soil surface after forest fires has been mentioned recently as a possible cause (Mallik et al., 1984; Onda et al., 2008) or preventer (Cerda and

Doerr, 2008; Larsen et al., 2009) of soil sealing in burnt areas. In a field rainfall simulator experiment on a sandy loam and gravelly silt loam soils, Woods and Balfour (2010) found that ash clogged pores in the coarse textured sandy loam soil, consequently reducing infiltration rate, while it did not affect infiltration rate of the finer textured silt loam soil. The authors postulated that compared to the silt loam soil, ash particles can enter larger pores in the coarse textured sandy loam, reducing hydraulic conductivity, and consequently decreasing infiltration rate (Woods and Balfour, 2010). Conversely, in a rainfall simulator experiment Larsen et al. (2009) tested the effect of ash on similar textured soils and found that ash not only did not clog pores in the soil profile, but increasing ash thickness on both soils reduced runoff rate and total soil loss amounts compared to bare soil. The authors postulated that the ash cover acted as mulch which provided a temporary soil protection from raindrop impact and postponed the formation of a seal on the soil surface, until it was eventually eroded (Larsen et al. 2009).

In bare soils with low structural stability, the main factor that controls soil infiltration rate under rainfall conditions is seal formation at the soil surface (Morin et al., 1981; Ben-Hur et al., 1985, 1987; Assouline and Mualem, 2000; Ben-Hur 2008). This seal is only a few millimeters deep, dense and strong, with finer pores and lower saturated hydraulic conductivity than the underlying soil (McIntyre, 1958; Chen et al., 1980; Shainberg and Levy, 1996). Usually, soil seals are classified in two types: structural seals, which are formed due to raindrop impact and fast wetting; and depositional seals, which are formed by the sedimentation of particles carried by runoff (Bresson and Boiffin, 1990; Assouline, 2004). Structural seals are created by the following processes: (i) physical disintegration of soil aggregates caused by the impact energy of the raindrops and slaking. Slaking occurs when aggregates cannot withstand the stress produced by forces such as: differential swelling, compression of entrapped air, rapid release of heat during wetting, and mechanical action of moving water (Emerson, 1977; Collis-George and Green, 1979; Kay and Angers, 1999). The slaking process is controlled by the wetting rate of the soil: the faster the wetting rate, the stronger the slaking forces (Rengasamy and Olsson, 1991). When soils that are initially dry are exposed to rainfall, soil aggregates at the soil surface are wet fast, and therefore aggregate disintegration during a rainstorm could involve extensive slaking processes (Lado et al., 2004a).; (ii) soil compaction and realignment of soil particles by raindrops; and (iii) physicochemical dispersion of soil clays (Agassi et al., 1981; Shainberg and Letey, 1984, Assouline, 2004). Physicochemical dispersion of clay particles can cause their

movement into regions of reduced porosity, where they can clog water-conducting pores in the soil profile. Clay dispersion occurs when the electrolyte concentration of the suspension is below the flocculation value of the clay (Van Olphen, 1977; Shainberg and Letey, 1984), and it is directly related to soil properties such as clay content and Exchangeable Sodium Percentage (ESP), and to the properties of the soil solution such as EC, SAR and dissolved organic matter (Shainberg and Letey, 1984; Frenkel et al., 1992; Tarchitzky et al., 1999; Lado and Ben-Hur, 2009). Forest fires can increase the risk of seal formation as a result of changes in chemical properties of the soil (e.g., destruction of OM), which can lead to more slaking and clay dispersion when the soil is wetted. Furthermore, the absence of a surface cover leaves already vulnerable aggregates exposed to direct raindrop impacts, increasing detachment and the amount of soil particles that can clog the pores and form a seal.

2.2 Post-fire erosion mitigation treatments

Post-fire erosion mitigation treatments are designed to minimize the risk of runoff generation and soil erosion through physical control. The most effective treatments will be those that help maintain rainwater infiltration and prevent detached soil particles from leaving the slope (Robichaud et al., 2010). The effectiveness of hillslope rehabilitation treatments depends on the actual rainfall amount and intensities – especially in the first, second and third year after the fire (Robichaud, 2005). There are several techniques used for post-fire soil rehabilitation: (1) **Seeding**. As an emergency treatment was found to be the most cost-effective but has low potential for reducing soil erosion during the first season after the fire, since the benefits of the seeded grasses (providing a canopy that protects the soil against raindrop impact) minimize after the harmful initial rainfall events (Robicahud et al., 2010). (2) **Mulching.** This method consists on spreading material on the soil surface to protect it from direct raindrop impact and to reduce overland flow. Many materials are used as mulch, including paper, wood-chips, wheat and rice straw, jute, and natural and synthetic fabrics. Straw mulch was proven highly effective in reducing soil erosion rates after wildfires, with reductions between 50-94%, (Robichaud, 2005), relatively low cost and a low failure risk (Robichaud et al., 2010). Some mulch treatments are applied combined with seeding, in order to improve the germination of seeded grasses, preventing a decrease in water infiltration and enhancing soil water retention (Robichaud et al., 2005). A special type of mulch is hydromuching, a generic name for a combination of tackifiers, polymers, bonded fiber, seeds, and other proprietary ingredients that, when mixed with water

and applied to the soil surface, form a matrix that can reduce erosion and foster plant growth. Although proven very effective, large scaled applications of hydromulch are expensive compared to dry mulching (Robichaud, 2005) and require substantial amount of water, a scarce resource in semi-arid and arid countries. (3) **Log erosion barriers** (LEBs). Also known as contour-felled logs or log terracettes, involves laying logs on burned-over hillslides in a direction perpendicular to the slope, in order to create mechanical barriers to water flow, promoting infiltration, trapping sediment and restricting soil movement. However, this method is costly and its effectiveness has been found to be contradictory (Robichaud et al., 2010), with some studies e showing negative effectiveness, others showing null effect, and others delivering positive results compared to untreated hillsides (Robichaud , 2005). Other rehabilitation methods such as Contour trenching and Straw wattles, are being evaluated nowadays (Robichaud, 2005; Cerdá and Robichaud, 2008).

2.3 Synthetic polymers to control soil erosion

2.3.1 Introduction to polymers

An alternative method to protect the soil against structural degradation and soil erosion is the application of substances that modify soil properties involved in seal formation, runoff generation and erosion. Among them, the most popular are synthetic polymers that improve soil physical properties (as reviewed by Levy and Ben-Hur, 1998; Ben-Hur, 2006, and Sojka et al., 2007). There are several types of polymers used as soil conditioners such as polysaccharides (PS) and polyacrylamides (PAM), being the last ones the most widely used to control soil erosion. PAM consists of repeated small identical units of carbon based monomers covalently coupled together to form extended chains. The length of the PAM chains depends on the number of monomers in the molecule and can reach up to a few million monomer units (Schamp and Hoylebroek, 1973). The PAM molecule length in solution ranges between a few micrometers to several hundred, their average diameter is 0.5-1.0 nm, and their molecular weight ranges from a few thousands to tens of millions of daltons (Schamp et al., 1975). Similar to other polymers, PAM chains are flexible and have a changing conformational structure containing functional groups (Seybold, 1994). These groups can be cationic, anionic or nonionic. Increasing the hydrolysis rate of the PAM causes it to stretch its conformational structure due to repulsion between charges of the same sign located in the functional groups (Malik and Letey, 1991). The more stretched the PAM molecule is, the more it gets a stick-like conformation, and when less

stretched it coils and gets a more ball-like structure (Michaels, 1954). PAMs are usually characterized by their molecular weight (MW), degree of hydrolysis, molecular conformation (coiled or stretched), type of charge and charge density (Levy and Ben-Hur, 1998; Ben-Hur, 2006). In this work we focused on testing the effectivity of anionic PAMs as a post fire erosion control method, hence the introduction will concentrate on this specific type of polymer.

2.3.2   PAM-soil interaction

The presence of functional groups, especially in anionic polymers, allows the development of strong interactions of polymer with soil particles, especially with clay. Anionic PAM get adsorbed to clay particles in three cases (Greenland, 1972, Theng, 1982; Ben-Hur, 2006): (i) when interaction between the negatively charged functional groups of the PAM and the positively charged groups of the clay minerals occur in acidic pH; (ii) when the negative charge on the PAM is screened by high electrolyte concentration; and (iii) when exchangeable polycations act as 'bridges' between the negative charge of the PAMs' functional groups and the negatively charged planar surface of the clay, in what is termed 'bridging adsorption'. The affinity of PAM to clay depends on: (i) the type and density of the electrical charge of the PAM; (ii) the structural conformation of the PAM, (iii) clay properties; and (iv) the environmental conditions in the soil such as pH and the electrolyte content of the soil solution (Greenland, 1972; Theng, 1982).

2.3.3   Effect of PAM on runoff and soil loss

Nowadays, the addition of high molecular weight anionic PAM to the soil surface in order to reduce soil erosion is a common method widely used in furrow and sprinkler irrigation (Levy et al., 1991; Trout et al., 1995; Aase et al., 1998; Santos et al., 2003; Sojka et al., 2007; Lentz and Sojka, 2009; Lentz and Westermann, 2010). Besides PAM and soil characteristics, the efficiency of the polymer to stabilize soil structure depends on environmental factors such as electrolyte concentration in the soil solution (Shainberg et al., 1990), and soil drying after the polymer application. During drying, soil particles and PAM molecules come closer, which causes an increase in the number of contact points between them, and result in an enhancement of soil stabilization (Shainberg et al., 1990).

Levy et al. (1991) studied the effect of spraying anionic PAM prior to the irrigation season at a rate equivalent to 20 kg ha^{-1} on runoff and soil loss of two soils (a clay loam loess and a silt loam vertisol), under high intensity irrigation via Moving Sprinkler Irrigation System.

17

The effect of PAM was checked on bare soil and soil covered with a crop. The results showed that PAM was significantly efficient in reducing seal formation and runoff generation in both soils, whether bare or covered with crops. The sediment concentration in the runoff was similar in both treatments, suggesting that PAM reduced erosion by reducing runoff, in addition to strengthening the resistance of the aggregates at the soil surface against drop impact (Levy et al., 1991). Additionally, PAM dissolved in irrigation water (10 mg L^{-1}) has been shown to prevent erosion and increase infiltration in furrow irrigation (Trout et al., 1995). Similarly, applying PAM in concentrations of 5, 10 and 20 mg L^{-1} to the irrigation water was proven to be effective in reducing runoff and soil erosion from a loam loess and a clay vertisol under sprinkler irrigation (Levy et al., 1992).

Despite of its proven effectiveness to reduce runoff and soil loss, dissolving PAM in aqueous solutions is difficult, and therefore the application of PAM in granular form should be more appropriate and economical, especially in dry regions where water is a scarce resource. Yu et al. (2003) pointed that using aqueous PAM is not economically efficient due to the large amount of time necessary to dissolve the polymer at high concentrations. Moreover, PAM/water solutions are very viscous and can overpower the equipment used for its application. Yu et al. (2003) analyzed the effect of spreading phosphogypsum (PG) and granular high–MW anionic PAM (12x10^6 Da, 15% hydrolysis) on IR and erosion in two soils (a silty loam and sandy clay) exposed to distilled water (DW) rain. The treatments were: (i) control, (2) granular PAM mixed with soil at a rate of 20 kg ha^{-1}; (iii) two rates of PG spread on the surface; (iv) two rates of granular PAM combined with two rates of PG. While applying a combination of PG and PAM resulted in a significant increase in IR and a decrease in soil loss, spreading granular PAM alone reduced IR substantially. Yu et al. (2003) suggested that dissolved PAM adsorbed to soil particles in the external part of the aggregates are long and block the water conducting pores. Therefore, the authors assumed that the reduction in IR is due to a reduction in soil hydraulic conductivity rather than soil sealing. However, in the application of dry PAM alone, soil loss was lower compared to all the other treatments, decreased almost 3 fold in the sandy clay soil, and about 6 times in the silty loam loess soil, compared to the control treatments. These authors postulated that the reduction in soil loss is a consequence of the adsorption of the long PAM molecules on the soil surface, binding soil particles together.

2.3.4 Stabilizing steep slopes with PAM

The application of PAM has been also been successful in stabilizing unconsolidated soils in steep slopes. Flanagan et al. (2002) tested the effect of spraying 80 kg ha^{-1} anionic PAM with and without gypsum (5 Mg ha^{-1}) on soil loss from a clay loam in a highway cutslope (35% slope) and a silt loam in a landfill (45% slope), exposed to natural rainfall. The PAM treatments significantly decreased soil erosion in both sites, reducing the sediment yield between 40-54% compared to the control. The addition of gypsum did not have a significant effect on runoff volume in one of the sites possibly because this soil already contained significant amounts of Ca. Flanagan et al. (2002) also found that the application of PAM and PAM/gypsum increased grass establishment and growth compared to the control soils.

2.3.5 Mechanisms of stabilization of soil structure

The effectiveness of PAM to maintain soil structure is often related to its ability to promote flocculation (Sato and Ruch, 1980; Aly and Letey 1988; Helalia and Letey, 1988). The flocculating action of the polymer is derived from its ability to interact with several clay tactoids at the same time, creating assemblages of bigger size (floccules) that flocculate. The flocculation of dispersed clay mediated by a conditioner could be a result of (Ben-Hur, 2006): (i) neutralization of the repellant forces between clay particles by the electrostatic adsorption of polymer molecules to the clay; and (2) bridging of soil particles. High molecular weight anionic PAMs are effective flocculants, especially in the presence of polyvalent cations (Roberts et al., 1974; Gu and Doner 1992) due to their long grappling distance that facilitates interparticle bridging (Theng, 1982). Ben-Hur and Keren (1997) studied the capability of anionic, cationic and nonionic PAMs to form aggregates in suspensions of a sandy loam soil containing 0.1% CaCO$_3$ and with ESP of 1.9%. The results showed that anionic PAM was the most effective polymer to promote aggregate flocculation. Shainberg and Levy (1994) found that the efficiency of anionic polymers in preventing seal formation is enhanced when the soil clay is maintained in a flocculated state. An increase of electrolyte concentration causes flocculation of clay particles, because it slows the physical disintegration of soil aggregates and chemical dispersion of clays (Shainberg et al., 1990; Tang et al., 2006). Smith et al. (1990) found that treating three Israeli soils covering a range of clay contents with a combined treatment of gypsum and anionic PAM was more effective than each of the treatments by itself increasing IR and reducing erosion.

The adsorption of PAM to clay is affected by the structure of clay particles and of floccules (micro-aggregates) that the interactions PAM-tactoids form (Ben-Hur et al., 1992). Clay particles in suspension usually exist as individual clay platelets arranged parallel to each other therefore forming small groups called tactoids (van Olphen 1977). Ben-Hur et al. (1992) studied the adsorption of PAM with various charge densities to Ca-illite and Ca-montmorillonite tactoids in suspension, and they observed that adsorption of PAM on both illite and montmorilonite tactoids took place on their external surfaces, since the interlayer space is not big enough to allow the penetration of PAM. Moreover, when flocculated with sufficient amount of PAM or electrolytes, Ca-illite and Ca-montmorillonite tactoids formed floccules with relatively small pores. Ben-Hur et al. (1992) observed that adsorption of PAM to floccules was bigger in Ca-illite than Ca-montmorillonite, since Ca-illite floccules tended to have bigger pores than the ones of the Ca-montomorillonite (O'Brien, 1971), which allowed the penetration of relatively long coils of PAM into intra-aggregate spaces.

2.3.6 Coating or penetration into aggregates

There are some contradictory findings regarding whether aggregate stabilization by PAM molecules occur from the inside (PAM penetrates the aggregates) or from the outside (PAM locates on the exterior surface of the aggregates). Malik and Letey (1991) measured the adsorption of various types of anionic PAM on several Mediterranean soils (Arlington, Bosanko and Fallbrook) with clay content ranging between 10-20%. The results showed that soils with the same aggregate size but different surface area (as a result of different clay type and content and same low exchangeable sodium percentage-ESP) had the same adsorption values for each PAM type. The researchers suggested that PAM did not penetrate into the aggregates but was adsorbed mainly on their external surfaces. On the other hand, soils with high ESP (34%) showed high PAM adsorption rates, probably because clay swelling at this ESP resulted in higher surface area that promoted PAM adsorption. Levy and Miller (1999) found that pores in small-size aggregates (<1 mm) are narrow and may not allow the large molecules of high MW PAM to penetrate, while larger aggregates, although they are also stabilized from the outside, have larger pores big enough to allow the PAM to penetrate, adsorb to internal surfaces and create inner stabilization. Similarly, Mamedov et al. (2007) studied the effect of PAM application on the stability of aggregates from four smectitic soils varying in their clay content (22.5- 65%). The penetration of PAM into the aggregates was estimated by treating with PAM two aggregate sizes: 0.5- 1.0 and

1.0- 2.0 mm and comparing the stability of the small aggregates with that of the large aggregates after crushed and sieved to 0.5- 1.0 mm size. These researchers found that in the fine textured soils, the stability of the initially small aggregates was generally greater than that of the initially large aggregates, indicating that most of the PAM was adsorbed on the exterior surfaces and only a small fraction of it, if any, entered into pores.

2.3.7 Desorption of PAM

In addition to adsorption of PAM molecules to soil particles, an important factor that influences the capability of PAM to stabilize soil structure is desorption from soil particles, which indicates the prevalence of the effect of the polymer in bridging soil particles. Nadler et al. (1992) tested desorption of tritium labeled PAM and PS from washed quartz sand and three Israeli soils. Soils were saturated with deionized water (DW) after being pretreated with tritium labeled polymer solution, and the amount of tritium in the water was measured after shaking and centrifugation. The results showed that PAM desorption was negligible or very low when the soils remained wet, and if the soil is dried, most of the PAM becomes irreversibly bonded to the soil.

2.3.8 PAM interactions with cations in the soil solution

The presence of divalent cations in the soil solution enhances the positive effect of PAM on infiltration and hydraulic conductivity (Shainberg et al., 1990). Shainberg et al. (1990) tested the efficiency of sprayed anionic PAM to reduce the decrease in IR of two soil types under simulated rainfall using both distilled- or tap-water. These authors found that increasing the electrolyte concentration in the soil solution by the addition of phosphogypsum or using tap water increased the efficiency of PAM in stabilizing aggregates, preventing soil sealing and increasing IR. Shainberg et al. (1990) hypothesized that the increase in the effectiveness of PAM is a result of a "bridging effect" mediated by polycations between the negatively charged PAM and the negatively charged clay minerals. These authors also suggested that when the concentration of electrolytes in the soil solution exceeds the flocculation value of the clay, the cementing action of PAM is more effective (Shainberg et al., 1990).

2.3.9 Viscosity

Malik and Letey (1992) found that when soil is leached with high MW PAM solution, the presence of PAM increases the viscosity of the solution, which leads to a reduction of soil hydraulic conductivity. Ben-Hur and Keren (1997) found that the viscosity of anionic PAM (GC;

1 x 10^7 Da; 20%CD) was higher than the viscosity of smaller nonionic (P-101, 1-2 x 10^5;<2% CD) and cationic (CP-14, 2 x 10^5-2 x 10^6Da; 10% CD) polymers. In a rainfall simulation experiment, the anionic PAM showed the smallest efficiency in increasing IR, which was also attributed to its high viscosity values. The authors ascribed the high viscosity of the anionic PAM to its high MW and charge density (Ben-Hur and Keren, 1997). The viscosity of the PAM solution depends on several properties: (i) the conformational structure of the molecule (Malik and Letey, 1992); (ii) the MW of the PAM (Volk and Friedrich, 1980); (iii) the concentration of PAM in the solution (Malik and Letey, 1992); (iv), the type of charge and charge density of the PAM (Malik and Letey, 1992); and (v) the electrolyte concentration of the soil solution (Malik and Letey,1992; Yu et al., 2003; Ajwa and Trout 2006).

Yu et al. (2003) tested the effect of applying PAM (A110, anionic PAM, 12x10^6Da and 15% Hydrolysis) with different applications and rates on runoff and soil loss yield from silty loam and sandy clay soils under a rainfall simulator. The PAM applications were control (i.e no PAM), 20 kg ha^{-1} dry PAM, two rates of gypsum (2 and 4 Mg ha^{-1}) and PAM mixed with gypsum (PAM 10- and 20- kg ha^{-1}, gypsum 2 and 4 Mg ha^{-1}). In both soils, the dry PAM treatment showed the lowest final infiltration rate (FIR) and high runoff amounts, similar to those found in the control treatment. Nevertheless, in spite of the increase in runoff, the addition of dry granular PAM alone yielded the lowest soil loss from all the aforementioned treatments.

Ajwa and Trout (2006) tested the effect of two types of high MW anionic PAM: Pristine (emulsified, 35% CD) and Superfloc A110 (granular; 18% CD), both with MW 12-15 Da, on the ability to flocculate soil particles in suspension and on infiltration rate on a sandy loam soil. For both PAM formulations, PAM ability to flocculate increased with its concentration. For all PAM concentrations, the emulsion PAM (Pristine) was less effective than the granular formulation (Superfloc A110) to produce flocculation of soil particles. Addition of Ca^{+2} originated in gypsum, improved flocculation in both formulations presumably due to the bridging effect of divalent cations on PAM molecules and clay particles. Furthermore, column infiltration studies demonstrated that water treated with emulsified PAM could reduce infiltration rates in sandy loam soils by more than 50%, and granular PAM by 75%, due to the increase of viscosity of the solution. Moreover, the reduction of soil infiltration was greater as PAM concentration was high, when PAM was applied continuously throughout the irrigation period, and when the solution contained low salt or high sodium concentrations. Ajwa and Trout (2006)

postulated that this reduction in infiltration was the result of a large increase in the apparent viscosity of PAM solution.

Increasing the electrolyte concentration of the solution reduces its viscosity (Yu et al., 2003; Ajwa and Trout, 2006) The presence of electrolytes cause the PAM molecules to coil up as a result of the reduction of repulsive forces between functional groups with the same charge inside the molecule, and the effect of the polycations bridging those functional groups. The resulting solution is formed by less stretched PAM molecules, and thus they are less prone to clog pores in the soil profile (Yu et al., 2003; Ajwa and Trout, 2006). On the other hand, the coiling of the PAM may reduce its effectiveness in stabilizing soil structure, since the "grappling distance" of the coiled molecule is shorter than in stretched conformation (Mamedov et al., 2007).

2.3.10 Effect of seedling emergence and biomass

Several studies have shown that PAM can be successful in improving seedling emergence rates in grass and other plant species (Cook and Nelson, 1986; Rubio et al., 1989). Flanagan et al. (2002) found that a steep road embankment exposed to direct rainfall treated with sprayed anionic PAM at a rate of 80 kg ha^{-1} experienced an increase in grass establishment and growth compared to a control treatment.

Soil sealing can restrict seedling emergence and therefore reduce crop yield since seals can be 200 to 2000 times harder than normal soil (McIntyre, 1958) and can be an impenetrable barrier to emerging seedlings (Chen et al., 1980). Therefore, the use of soil amendments that prevent surface sealing, such as PAM, can help seedling emergence and avoid the reduction of yield (Cook and Nelson, 1986; Wallace, 1987; Stern et al., 1992; Rapp et al., 2000). Stern et al. (1992) studied the effect of the application of high MW anionic PAM on runoff and wheat yield productivity under sprinkler irrigation experiment, in semi-arid conditions. PAM was sprayed at a rate of 20 kg ha^{-1} over dry soil following a 5Mg ha^{-1} PG application. The results showed that PAM application was highly beneficial for the crop, reducing runoff, and increasing soil water content and wheat biomass production. These authors suggested that the increase in crop yield was a result of the prevention of seal formation, which allowed more irrigation water to penetrate into the soil (Stern et al,. 1992). Similar results were obtained by Rapp et al. (2000) using rainfall simulations. In this case, the application of 20 kg ha^{-1} PAM with 5 Mg ha^{-1} PG was very effective in increasing the union emergence of onion seeds in a loess soil with low ESP.

However, this positive effect decreased when the ESP of the soil increased. Cook and Nelson et al. (1986) conducted a field experiment to test the effect of the application of granular and dissolved high MW anionic PAM on seedling emergence of soils prone to seal formation. Corn seeds were spread using a seeder before the PAM applications. An improvement in seedling emergence was observed when the soils were treated with PAM solutions, but not when treated with granular PAM. In another study, Steinberger and West (1991) found that mixing a soil with granular anionic PAM (Complete green) did not improve the emergence of the annual grass *Bromus tectorum* compared to a control treatment, although the latter had shown crust formation.

Contrary to most agricultural fields, forests occupy steep slopes, where erodibility is higher. Despite its potential beneficial effect, few studies have focused on using PAM to reduce soil erosion after wildfires. Davidson et al. (2009) conducted a three year experiment in which they compared the effects of the application of PAM and straw mulch on soil erosion and vegetation recovery. The experiment took place after the Red Bull Fire, Utah in July and august 2004, after a 67 ha^{-1} area have been seeded with local vegetation. Four treatments (covering an area of 8 ha^{-1}) were conducted in blocks: (i) Control; (ii) PAM; (iii) PAM+straw mulch; and (iv) straw mulch on slopes of 16%, 33%, 25% and 20%, respectively. The PAM was spread as recycled paper pellets containing PAM with an average MW of 18×10^6 Da, at a rate of 8 kg ha^{-1}. The results showed less soil erosion from the PAM and PAM+mulch treatments compared with just mulch and control treatments, although the differences were not significant. Davidson et al. (2009) explained that although differences were not significant, PAM and PAM+mulch treatments were on the steepest slopes, hence their soil erosion rates were expected to be the highest. PAM and PAM+mulch treatments showed a significant increase in the vegetation recovery in the first and second year after the fire compared with the other two treatments.

2.4 Scientific hypothesis of this work

1. Different fire conditions make different changes in soil properties which can affect the soil structure and by that alter the rainfall infiltration regimes.

2. According to previous success in agricultural and construction activities, PAM application can reduce soil loss in forest soils after various fire severities.

3. Different PAM applications after forest fire conditions do not affect aboveground growth and biomass and seed production.

2.5 Research objectives

Despite the proven effectiveness in reducing soil loss in agricultural fields, few studies have investigated the effect of PAM as a post-fire soil protection method against soil erosion. Therefore, the objectives of the present work were:

1. To study the post-fire changes in soil properties that affects the infiltration/runoff/soil loss relationships, in soils under different fire conditions.

2. To study the effects of anionic PAM application in soil erosion reduction in fire-affected soils under laboratory rainfall simulator and field conditions with natural rainfall.

3. To study the effect of different PAM application rates on vegetation recovery under natural conditions.

4. To evaluate the use of PAM as a tool for post-fire forest recovery

2.6 Research questions

1. What soil chemical and physicochemical properties are altered as a consequence of different fire conditions.

2. How does changes in soil chemical and physicochemical properties affect the infiltration/runoff/soil loss interactions, and to which extent?

3. Does the application of PAM help in the reduction of soil loss in disturbed soils by different fire severities?

4. What are the mechanisms in which PAM reduces soil loss, especially after several wetting/drying cycles?

5. How does PAM affect runoff and soil erosion after forest fire in natural conditions, and in different application rates in a burnt forest?

6. Which application rate is the most effective in reducing soil erosion after forest fire?

7. Does PAM affect aboveground biomass and seed production?

8. Can PAM be recommended as a tool for post-fire soil control treatments?

3 Materials and Methods

3.1 Soil samples and analysis

Two soil types were used in this study: (i) A Pale rendzina (Calcareous montmorillonitic sandy clay loam; Lithic Xerothenth; Singer, 2007) collected from Birya forest, in a planted (with *Pinus halepensis* and *Pinus brutia*) forest stand under Mediterranean climate; and (ii) a non-calcareous sandy loam soil from Barbanza, Galicia, Spain, was sampled under Maritime pine (*Pinus pinaster*) trees under oceanic climate. The soils were air-dried and crushed to pass a 2 mm sieve before performing soil analysis. Some of their physical and chemical properties are presented in Table 1. The particle size analysis was determined with a hydrometer (Day 1956) after oxidation of OM with hydrogen peroxide; OM content was measured using the Walkley-Black method (Allison 1965), $CaCO_3$ content by a volumetric method (Allison and Moodie 1965), and the cation exchange capacity (CEC) by extraction with ammonium acetate at pH 7 (Chapman 1965). Exchangeable sodium percentage (ESP) was determined by measuring the cations displaced by the ammonium acetate and calculated using the following Eq. [1]:

$$ESP = \frac{E_{Na} \times 100}{CEC} \qquad [1]$$

where E_{Na} is exchangeable Na^+ fraction (meq $100g^{-1}$) and CEC is Cation Exchange Capacity (meq $100g^{-1}$)

Some parameters of the soil solution composition were measured after batch extraction. In this procedure, 0.05kg of <4mm soil samples were mixed with 75 mL of deionized water (DW) in 200 mL Teflon centrifuge tubes. The tubes were sealed with Teflon lined cups, and shaken mechanically for 1 h at 160 rpm. Following the shaking process, the tubes were centrifuged for 10 minutes at 7000 rpm and the supernatant was collected. The pH values were determined by a pH meter; electrical conductivity (EC) measurements were performed using an EC meter; the concentrations of Na^+ and K^+ were determined using a flame photometer 243; the concentration of Ca^{2+} and Mg^{2+} was determined by standard titration method; and DOC was determined using a TOC-V CPN Total Organic Carbon Analyzer by Shimadzu. Sodium adsorption ratio (SAR) was calculated using Eq. [2].

$$SAR = \frac{[Na^+]}{\sqrt{\frac{1}{2}[Ca^{2+} + Mg^{2+}]}} \qquad [2]$$

where cation concentrations are expressed in $(meq\ L^{-1})^{-0.5}$..

3.2   Fire severity experiments

Forest fires have different effect on soil properties due to spatial variability in fuel load, soil properties and the duration of exposure to heat. These differences may influence the underlying soil differently, resulting in a mosaic pattern of different effects (Kutiel et al., 1995; Lavee et al., 1995). In order to evaluate the effect of different fire conditions, that may exist within a wildfire, on soil chemical and physical properties including IR, runoff and soil loss, three fire severity treatments were prepared. The experiments were conducted with the Pale Rendzina soil from Birya forest (see chapter 3.3.2.1). Three treatments were tested:

(i)   Unburned soil treatment represents an area that was not affected by fire – will be referred to as UB soil

(ii)   Soil affected by a prolonged exposure to moderate temperatures treatment represents a soil that was exposed to heat but was not exposed directly by the fire and did not have ash deposits on top of it – will be referred to as HT soil. Similar conditions can be found in a wildfire, when long lasting smoldering occurs and heat is radiated downwards towards the soil without direct contact with the flames (Neary et al., 2005).

(iii)   Soil affected by direct fire represents a soil that was exposed directly to wildfire, and had ash deposits on top of it – will be referred to as DF soil.

The UB and DF soil were collected in the 28th of October (2009), from the top 3 cm of the mineral soil in Birya forest (see chapter 3.3.2.1). Samples were collected from outside (UB) and inside (DF) a burnt area (see chapter 3.3.2.1) after carefully removing the duff or ash cover, respectively. The samples were carefully taken to the laboratory, air-dried, crushed, rock and gravel fractions were removed, and the remaining material was sieved through a 4mm-mesh sieve. Both soils were sampled at the end of November, 2009, after some rain events that occurred in September-November summing up to 214 mm of rain with three relatively intense rainstorms.

The HT soil was created in the laboratory. Part of the sieved UB soil was placed in a muffle for 8 hours at a temperature of $300^{\circ}C$. Naturally, forest soils rarely get exposed to high temperatures for such a long time unless slash burning takes place, which causes smoldering for hours or days (DeBano et al., 1998). However, with this procedure we tried to simulate changes in soil properties that can occur in the upper few centimeters of the soil, which are exposed to

Table 1: *Mechanical composition, organic matter (OM), CaCO$_3$ content, exchangeable sodium percentage (ESP), cation exchange capacity (CEC), and soil solution pH, electrical conductivity (EC), sodium adsorption ratio (SAR) and dissolved organic carbon (DOC) from the top 3 cm of the two soils used in this study.*

	Mechanical composition							Soil solution composition (1:1.5)			
	Clay	Silt	Sand	OM	CaCO$_3$	ESP	CEC	pH	EC	SAR	DOC
	---------------------------- % ----------------------------						meq 100 gr^{-1}		dS m^{-1}	(meq L^{-1})$^{0.5}$	mg L^{-1}
Pale Rendzina	26	23	51	6.5	61.4	1.2	20.9	7.3	0.4	0.1	294.4
Barbanza sandy loam	13	18	69	12.1	0.5	0.5	26.3	3.6	0.5	-	-

high temperatures during the fire and whose properties can reach a steady state in a relatively short time. These upper centimeters of transformed soil have a great impact in runoff generation and soil loss.

3.2.1   Effect of fire severity on infiltration, runoff and soil loss – Rainfall simulator study

The disturbed UB, DF and HT soil samples were packed in perforated trays measuring 0.30 m by 0.50 m and 0.02 m deep, with bulk densities of 1.2, 1.1 and 1.0 Mg m^{-3}, respectively. The trays were placed on an 8-cm-thick layer of crushed and washed shells, in a box positioned under rotary disc rainfall simulator (Morin et al., 1967) at a slope of 30%. The typical mechanical parameters of the simulated rainfall were: 1.9 mm raindrop mean diameter; 6.2 m s^{-1} median drop velocity; and 18.1J mm^{-1} m^{-2} kinetic energy. The duration of each simulated rainstorm was around 100 min, with a total of 80 mm of DW applied at a rainfall intensity of 47 mm h^{-1}.

Each soil was exposed to three 80mm rainstorms separated by drying periods in an oven at 37ºC during 72 hr. This procedure was conducted in order to imitate natural drying of the soils between rainstorms. The duration of the drying period was determined by introducing saturated UB soil trays in the oven and carefully weighing them each certain time until no changes in weight were evident (data not shown). A 30% slope was maintained in every rainstorm by measuring the slope with a spirit level, and carefully readjusting the slope if needed, before the onset of the rain.

Air-dry soils were exposed directly to the rainstorm in order to produce a relatively fast wetting of the aggregates at the soil surface at the beginning of the rainstorm, imitating natural conditions. Water percolating through the soil was collected and measured every 80 sec to determine the IR. The runoff generated during the rainstorm was collected in 4 fractions of 20 mm of cumulative rainfall each (~25 minutes), and measured at the end of the rainstorm. Total soil loss from each tray was measured by drying the runoff samples and weighing the dry material. The measured soil loss could be considered as interrill erosion, because the trays in the rainfall simulator were short (0.5 m) (Meyer and Harmon, 1984). Some of the leachate was collected during the rainfall in containers, which were replaced every 20mm of rainfall to create four 20 mm fractions in each rainstorm. Leachate and runoff fractions were left to rest (usually

for 12 hours) to allow fine particles to settle before taking 20 ml samples that were used to perform chemical analysis.

3.2.2 Aggregate stability tests

Aggregate stability tests were conducted in order to differentiate between the mechanisms that lead to changes in soil structure. Samples of UB, DF and HT soils were air-dried, visible roots organic residues and gravel were removed, and the dry samples were then sieved to obtain 2 groups of aggregates with size ranges of: <2 mm; and 2–4 mm. Several stability tests were performed according to Ben-Hur et al. 2009:

3.2.2.1 Slaking test

The aggregate stability of UB, DF and HT soils was determined, after wetting the 2–4 mm aggregates at each of 2 rates: fast wetting and slow wetting under vacuum.

(i) Fast wetting: 5 g of oven-dried (40°C) 2–4mm aggregates were immersed in a beaker containing 50mL of DW water. After 10 min, during which the aggregates remained at rest, the water was carefully removed under suction using a pipette. The soil fragments were transferred to a 50-µm sieve which had been previously immersed in ethanol, and were then gently moved up and down 5 times in ethanol to separate fragments <50 µm from larger ones. The >50 µm fraction was oven-dried and then gently sieved by hand through a column of sieves of mesh sizes 2.0, 1.0, 0.5, 0.25, and 0.1 mm. The weight of each fraction was measured and that of the <50-µm fraction was calculated as the difference between the initial weight and the sum of the weights of the other 6 fractions. The aggregate stability of each soil sample was expressed in terms of the mean weight diameter (MWD), which was calculated by using Eq. [3]:

$$MWD = \sum_{i=1}^{7} \overline{x}_i \, w_i \qquad [3]$$

in which w_i is the weight fraction of aggregates in the size class i with a mean diameter $\overline{x}_i$.

(ii) Slow wetting: 5 g of oven-dried (40°C) 2–4 mm aggregates were placed on a cotton cloth, the edges of which were immersed in DW water, inside a desiccator. The aggregates were wetted slowly under vacuum in the desiccator for 24 h. The wet aggregates were then transferred to a 50-µm sieve, which had previously been immersed in ethanol, and were then sieved as described above for the fast wetting test. The MWD values of the aggregates after the slow wetting

procedure were calculated by Eq. [3]. The slaking value (SLV) of each soil sample was calculated by using Eq. [4]:

$$SLV = \frac{MWD_s}{MWD_f} \qquad [4]$$

in which MWDs and MWDf are the mean weight diameters under slow and fast wetting conditions, respectively.

3.2.2.2 Dispersion test

The soil dispersivity was determined according to Gupta et al. (1984). A 2-g sample of each soil was suspended in 0.05 L of deionised water in a 0.5-L centrifuge tube. The tubes containing the suspension were shaken on a reciprocal shaker for 30 min at 20 rpm, and then immediately centrifuged at a relative centrifugal force of 960rcf for 5 min (J2-21M, Induction Drive Centrifuge, Beckman, USA). The concentration of dispersed clay in the turbid supernatant was determined by measuring the absorbance at 420 nm with a spectrophotometer (Thermo Scientific. Genesys 10uv), and comparing the result with a calibration curve of absorbance vs. suspended clay concentration that had previously been prepared for each soil type. The dispersion value (DV) for each soil sample was determined by using Eq. [5]:

$$DV = \frac{M_d}{M_t} \times 100 \qquad [5]$$

where M_d is the mass of the dispersed clay in the turbid supernatant per 1 g of tested soil sample, and M_t is the total clay mass in 1 g of tested soil sample.

3.2.3 Water repellency tests

Repellency persistence is typically measured by the water drop penetration time (WDPT) test (Letey, 1969), whereby the amount of time required for a given drop volume of deionized water placed on the surface of a soil sample to fully penetrate the soil is measured. The longer the WDPT, the more persistent is the repellency. WDPT test was conducted in the laboratory using aggregates < 2 mm.

3.3 Effect of PAM application on infiltration, runoff and soil loss

In order to test the effect of PAM on runoff and soil loss, two sets of experiments were conducted: laboratory rainfall simulation, and field experiments in runoff plots.

3.3.1 Laboratory experiments

3.3.1.1 Rainfall simulations

The rainfall simulation experiments were conducted similarly to the ones used to determine the effect of fire severity on runoff and soil loss (see chapter 3.2.1). Two soils were used in this set of experiments: the Pale rendzina collected from Birya forest, after the different fire severity treatments described in the previous experiments; and an oven burnt non-calcareous sandy loam soil from Barbanza (BSL) in Galicia, NW Spain.

Similar to the Pale rendzina soil, BSL soil that was used for the rainfall simulator PAM experiment was exposed to 8 hours of $300^{\circ}C$ in a muffle to simulate severe fire conditions on the upper few centimeters of the soil. This soil was used in order to compare the effect of PAM on a different soil (i.e., different parent material; almost no carbonates; different acidity etc.).

The polymer used in this work was anionic PAM (Superfloc A-110 Flocculent, manufactured by Kemira Water Solutions BV, Netherlands). The MW of the PAM is $1x10^{-7}$ Da and it has a 15% hydrolysis. The PAM was spread on the surface in granular form at a rate of 50 kg ha^{-1} (50 kg ha^{-1} PAM treatment). The 50 kg ha^{-1} rate was selected due to its best performance in reducing soil loss after several preliminary rainfall simulator experiments with different PAM rates that were conducted in the lab (data not shown). After spreading the polymer, each soil was exposed to three 80mm rainstorms with a rainfall intensity of 47 mm h^{-1}.separated by drying periods in an oven at 37ºC for 72 hr, as in the previous set of experiments. After each rainstorm, 20 ml samples were taken from each of the four fractions of leachate and runoff, and chemical analysis and relative viscosity measurements were conducted. The results from the 50 kg ha^{-1} PAM treatments were later compared to the corresponding control treatments with no PAM application (See chapter 3.2.1).

Relative viscosity measurements were conducted using an Ostvald glass tube viscometer by comparing the viscosity of the solution (leachate or runoff), with the viscosity of DI water. RV is usually calculated using Eq. [6]:

$$RV = \frac{\eta_s}{\eta_w}$$
[6]

where η_s is the viscosity of the solution; and η_w is the viscosity of DI water.

In each measurement, 0.015L of the leachate/runoff samples was placed in the viscometer. Timing measurements were performed by measuring the leachate/runoff solution passage time between two points. Relative viscosity was then determined by the calculating the ratio between the passage time of the leachate/runoff solution and the passage time of DI water.

3.3.1.2   Effect of PAM on aggregate stability

Aggregate stability experiments were performed in the laboratory with PAM-treated aggregates, in order to study the effect of PAM application on aggregate stability. Aggregates of 2-4 mm in size from the DF soil were pre-wetted with DW or a 1000 mg L^{-1} PAM-solution, for control and 50 kg ha^{-1} PAM treatments, respectively. The Pre-wetting was done by carefully exposing the aggregates to the PAM solution in a quantity equivalent to 50 kg ha^{-1} application. After the pre-wetting, aggregates were dried in an oven at 37°C for 24 hr and then carefully sieved again through a 2 mm mesh sieve in order to eliminate broken aggregates. With the remaining aggregates fast and slow wetting tests were performed as described in chapter 3.2.2.1. Dispersion tests were also performed with the same soils and with the same 1000 mg L^{-1} PAM-solution. Aggregates of < 2 mm in size were carefully pre-wetted with DW or PAM-solution, for control and 50 kg ha^{-1} PAM treatments, respectively. The pre-wetted aggregates were dried at 37°C during 24 hours. After drying, dispersion tests were performed as described in chapter 3.2.2.2.

3.3.2   Field experiments

3.3.2.1   Experimental site

The study area was located in Birya forest, in northern Israel (32° 59' 52N, 35° 30' 27N). Birya forest is one of the largest planted forest in the Galilee and is spread over 2000 ha: starts in the eastern hill slopes of Safed city towards Rosh Pinah and Hatsor Haglilit, spreads north until Dalton stream and reaches as far west as Amud stream. The average height of the forest is 840 m above sea level; with typical Mediterranean climate; the average annual temperature is 22°C; and the average annual precipitation is 600 mm. The study area was located in a planted pine forest stand in the outskirts of Birya forest (Fig. 1A), near Safed city (Fig.1A) dominated by Pale rendzina soil on top of marl and chalk sedimentary rocks. The forest stand was mainly made of a mix of Aleppo pine (*Pinus halepensis*) and Turkish pine (*Pinus brutia*), with a mix of old (>60 yr) and young trees. Due to intentional arson, some of the aforementioned forest area was burnt in a low-moderate fire (according to Pausas et al., 2003) in the 21st of July, 2009. During the fire,

the forest floor was burnt, the litter layer was scorched and black ash deposits were accumulated on the surface. The trunks of the older trees were scorched up to a few meters high and some of the younger trees were completely incinerated. Most of the trees still had leaves on their branches at the time when the experiment was set up, but most of them were brown and later fell on the forest floor. At the 11th of November 2009, an area of ~0.2 ha inside the burnt zone, was manually logged in order to imitate salvage logging post-fire treatment. Cut trees and branches were carefully and manually removed from the site without dragging them on the soil, to avoid disturbances on the forest soil.

3.3.2.2   Runoff plots

The field experiment was conducted in the burnt part of the aforementioned forest stand in Birya forest in the end of November 2009. The hillslope selected for the experiment had a westerly aspect and a uniform average 40% slope. As stated previously, before building the runoff plots, a total of 214 mm of rainfall precipitated in several events, but had almost no effect on the ash deposits on the soil floor. Twelve 4.5 m^2 runoff plots were built in the 22nd of November, 2009, in the clear cut area. Each plot was built using 200x15cm metal plates that were carefully inserted to the soil deep enough to prevent runoff exiting from the borders, but still preventing runoff and eroded soil from outside to penetrate into the plot (Fig. 1B). In the bottom of each plot, a funnel with a sediment trap was placed, and was connected to a 200L container located in the bottom of the slope by means of polypropylene tubes. The runoff generated in the plots would go down the slope, concentrate and pass through the funnel and accumulated inside the container. Ash deposits were carefully removed by hand from the soil surface before the application of PAM in order to isolate the effect of the PAM from other factors that could influence runoff and soil loss.

Three treatments, with four replicates each, were applied in randomized block design: (i) control treatment– no PAM was applied; (ii) application of 25 kg ha^{-1} PAM; and (iii) application of 50 kg ha^{-1} PAM. PAM was applied in dry granular form spread on the soil surface inside the plot, and 0.5m around the outside of the plot . Furthermore, after few rainstorms, herbicide was sprayed inside of every plot in order to avoid the growth of vegetation that could affect soil loss measurements.

Rain amount (mm) was measured with five rainfall gauges that were placed between the plots in the clear cut area. Several rainfall gauges were used in order to check if there are spatial
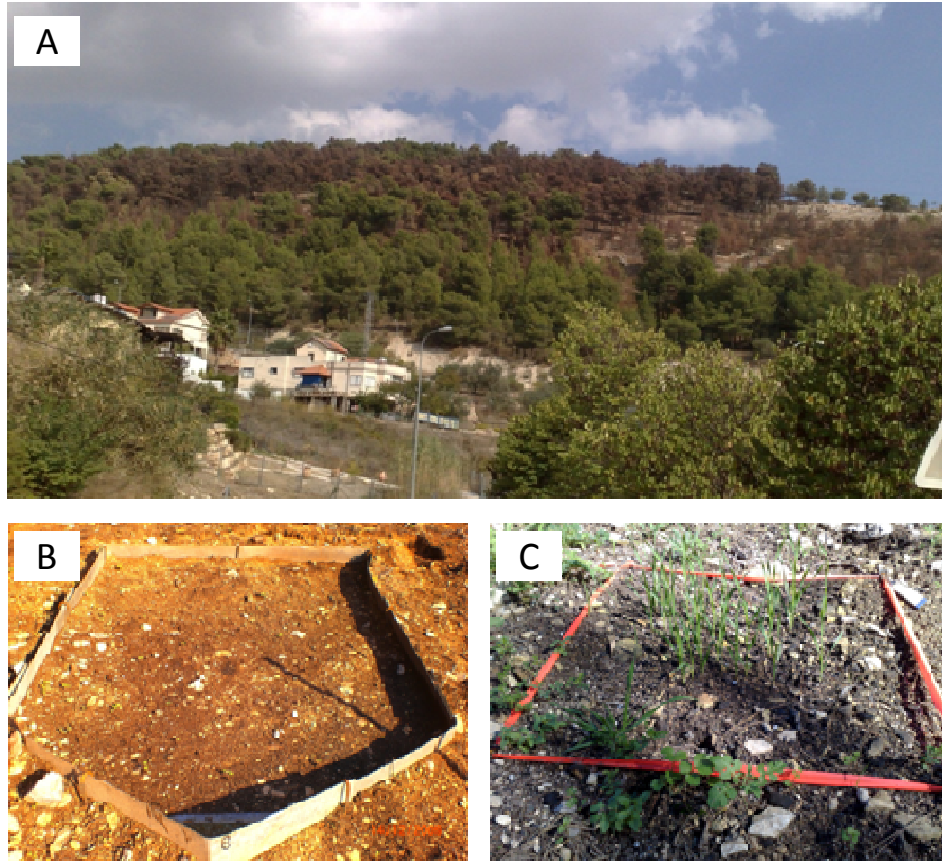
Fig. 1: *A. Burnt pine forest stand in the outskirts of Birya forest. The runoff plots were built in the middle of the brown area. B. an example of a 4.5 m² runoff plot constructed for the field experiment C. an example of a 0.25 m² wheat plot used to study the effect of PAM on the growth of annual grass after fire.*

differences in precipitation in each rainstorm around the area of the plots. No significant differences were found between the gauges (data not shown), indicating that the rain was spread evenly on all of the plots. Runoff and soil loss were measured after every rain event. All of the sediment generated from the plots was manually removed from the sediment trap in the funnel and weighed on a storm-by-storm basis. The values from individual storms were corrected for moisture content and normalized by the contributing area. From each container, 0.25 L runoff samples were taken for chemical analysis. In addition, 1 L samples were collected from the containers after mixing their content to determine the sediment concentration in the runoff. These samples were then dried in the laboratory in 105°C, weighed and multiplied by the total volume of the runoff for the specific rainstorm. The value was added to the sediment collected from the

sediment trap in the funnel of a specific plot, resulting in total soil loss for a specific rainstorm per plot.

3.4   Vegetation response to PAM

Fifteen square-shaped 0.25 m^2 plots were built in the burnt area next to the runoff plots to measure the vegetation response to PAM application (Fig. 1C). The plots were seeded with 30 seeds of wheat (*Triticum aestivum L.*) to simulate the development of annual grass vegetation after fire. PAM was applied after sowing using the same treatments than inside the runoff plots. Each treatment included five replicates. At the end of the winter, the plants inside each plot were cut near the root, put in paper bags and dried at 60°C for three days and weighted to calculate aboveground total biomass production. Furthermore, seed biomass production was measured weighing the seeds of all the plants in each plot.

3.5   Chemical analysis of leachate and runoff

Chemical analysis of the leachate and runoff samples from the rainfall simulator and runoff samples from the field experiments were conducted in the laboratory. These analysis included EC, pH, Na$^+$, Ca^{2+}+Mg^{2+} and DOC were analyzed as described in chapter 3.1.

3.6   Data analysis

Rainfall simulator and field experiments were conducted with four replicates per treatment in a complete randomized design. Numerical data was statistically analyzed using the software JMP 5.0.1 (SAS Institute, Inc.). The differences between the means and the interactions between the studied parameters with consecutive nature (rainfall simulator studies) were subjected MANOVA repeated measures model as a complete randomized design, using Wilks' Lambda test. The rest of the differences among the means were subjected to ANOVA model using Tukey's Significant Differences test (Steel and Torrie, 1981). All tests were performed at the α=0.05 significance level.

4   Results and Discussion

4.1   Part I: Fire effects on physical and physicochemical properties of the soil, surface runoff and soil loss

4.1.1   Results

4.1.1.1   The effects of fire severity on soil chemical and physicochemical properties

Fire treatments caused changes in the chemical and physicochemical properties of the Pale rendzina soil. The chemical and physicochemical characteristics of the UB, DF, and HT rendzina soils are presented in Table 2. A significant decrease in clay fraction of about 35% and in sand fraction of about 10%, along with increase in the silt fraction of 57%, was found in the HT treated soil compared to the UB soil, while no significant differences were found between the UB and the DF soil. Organic matter decreased significantly in the HT soil compared to UB soil (Table 2). Conversely, organic matter increased significantly in the DF soil. Furthermore, dissolved organic carbon was found to be significantly higher in the HT soil as compared to the other two soils (Table 2) while no significant differences were found between the dissolved organic carbon in the DF and UB soils (Table 2). ESP levels were relatively low in all soils (i.e., 1.2-, 1.1- and 1.9 % for UB, DF and HT soils, respectively); nevertheless, it significantly increased in the HT soil and decreased in the DF soil compared to UB soil. Consequently, cation exchange capacity (CEC) decreased significantly in the HT soil and increased significantly in the DF soil compared to the UB soil (Table 2). No significant differences were found between the pH of the three soils. The EC showed a significant increase in the HT soil, whereas it did not show a significant difference in the DF soil (Table 2). The SAR of the UB soil was significantly higher than the value in the burnt soils, whereas the HT soil had the lowest value of the two (Table 2). Carbonate content increased significantly only in the HT soil, while no significant differences were found between the DF soil and the UB one (Table 2).

4.1.1.2   The effects of fire on soil water repellency

Water repellency has been shown to be induced by burning (Doerr et al., 2009b), and it has been commonly cited as a primary cause of the observed post-fire increases in overland flows and sediment yields in some soils (DeBano, 1981, 2000; Robichaud, 2000; Shakesby et al., 2000). In the present work, a laboratory water drop penetration time (WDPT) experiment was

Table 2: *Mechanical composition, organic matter (OM), CaCO$_3$ content, exchangeable Sodium percentage (ESP), cation exchange capacity (CEC), and soil solution pH, electrical conductivity (EC), sodium adsorption ratio (SAR) and dissolved organic carbon (DOC) of the Pale rendzina soil after different fire treatments. Different lower case letters indicate significant differences between soil severity treatments within a soil property*

| | Mechanical composition | | | | | | | Soil solution composition (1:1.5) | | | |
	Clay	Silt	Sand	OM	CaCO$_3$	ESP	CEC	pH	EC	SAR	DOC
	----------------------------- % -----------------------------						meq 100 g^{-1}		dS m^{-1}	(meq L^{-1})$^{0.5}$	mg L^{-1}
UB	26 a	23 b	51 a	6.5 b	61.4 b	1.2 b	20.9 b	7.3a	0.4 b	0.14 a	294.4 b
DF	25 a	25 b	50 a	10.7 a	57.6 b	1.1 c	23.5 a	7.2a	0.3 b	0.1 b	163.9 b
HT	17 b	36 a	47 b	4.5 c	68.4 a	1.9 a	14.9 c	7.3 a	2.7 a	0.07 c	997.2 a

conducted on samples of the UB, DF, and HT soils. All three soils showed almost no water repellency, as their WDPT was <5 seconds (i.e. approximately 1 second for the UB and DF soils and even faster in HT soil, data not shown).

4.1.1.3   The effects of fire on aggregate stability

The mean weight diameter (MWD) of 2-4 mm aggregates of UB, DF and HT soils after fast and slow wetting, and their slaking values (SLV) determined using the slaking test (Eqs. [3] and [4]), are presented in Fig. 2. SLV represents the importance of slaking in aggregate breakdown. The lower the SLV, the more resistant the aggregates are to physical breakdown by raindrop impact and fast wetting (Lado et al., 2004a; Ben-Hur et al., 2009). In general, all soils showed high stability to slaking forces. No significant differences in MWD were found between the soils after slow pre wetting (significance not shown). However, the UB and DF soils had significantly lower MWDs after fast wetting, hence their slaking values were higher than those of the HT soil.
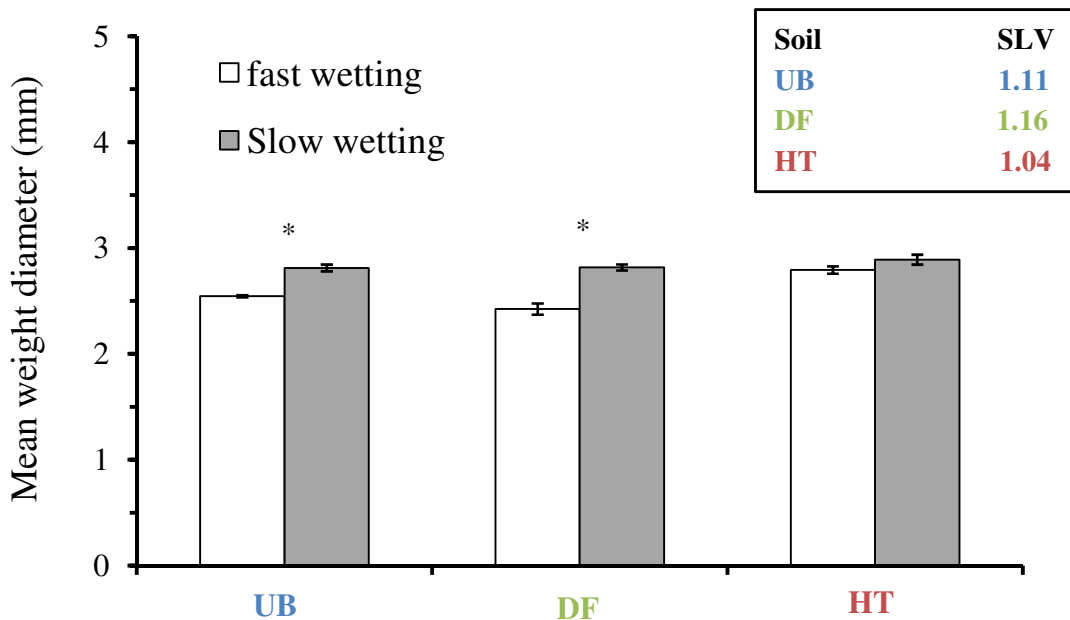

Fig. 2: *Mean weight diameter after fast or slow wetting for UB, DF and HT soils. An asterisk indicates significant differences between fast and slow wetting in each treatment (α≤ 0.05). Bars indicate one standard error.*

Dispersion values of UB, DF and HT soils and EC and SAR values of the solution during the test are presented in Table 3. The HT soil had the lowest dispersion value and the highest EC (7.7% and 0.35 dS m^{-1}, respectively), while no differences were found in the dispersion value

and EC of UB and DF soils (dispersion values of 12.3% and 11.5%, and EC values of 0.1 and 0.1 dS m^{-1}, for UB and DF soil, respectively) . The SAR of the solution of the three soils was low and similar, although the UB soil showed significantly higher value than the other two soils (Table 3).

Table 3: Dispersion value, electrical conductivity and sodium adsorption ratio measured in a dispersion test for the UB, DF, and HT soils. Difference letters indicate significant differences between treatments in each measured parameter.

Treatments	Dispersion value	Electrical conductivity	Sodium adsorption ratio
	%	*dS/m*	$(meq\ L^{-1})^{0.5}$
UB	12.3 a	0.1 b	0.05 a
DF	11.5 a	0.1 b	0.03 b
HT	7.7 b	0.35 a	0.03 b

4.1.1.4   The effects of different fire regimes on infiltration rate and soil erosion – rainfall simulator study

Infiltration is the entry of rain water into the soil. It is a dynamic process and it is one of the most important processes in the soil phase of the hydrological cycle since it determines the amount of runoff as well as the supply of water to the soil profile. Infiltration rates of UB, DF and HT soils as functions of cumulative rainfall, during three consecutive rainstorms separated by a 72 h drying period, are presented in Fig. 3. The data was collected during a laboratory rainfall simulation experiment. The onset of recordable IR in all soils and all rainstorms began after approximately 20 mm of rain, since the soil was initially dry and some amount of water was needed to saturate the soil layer.

In the 1st rainstorm, the IR of the UB soil decreased sharply, and was lower than that of DF and HT soils, ending in a final IR of 10 mm hr^{-1} (as compared to a final IR of 18.4- and 29.8 mm hr^{-1} for the DF and HT soils, respectively). In the same rainstorm, the IR of the HT soil was the highest between the two burnt soils (Fig. 3).

Generally, in the 2nd and 3rd rainstorm, the IRs of all the soils reached a final steady value after 20 mm of rainfall (Fig. 3). Nevertheless, the IR values of both DF and HT soils decreased from one rainstorm to the next, and the differences with IR values of the UB soil were smaller.

Soil loss collected from UB, DF and HT soils in four fractions in each rainstorm during the three consecutive rainstorms is shown in Fig. 4 as function of cumulative rainfall. In general, total soil loss increased in all the soils within each rainstorm (Fig 4). In most of the 1st and 2nd rainstorms, fractioned soil loss yields were in the following order: UB>DF>HT. However, while the HT soil still showed lowest
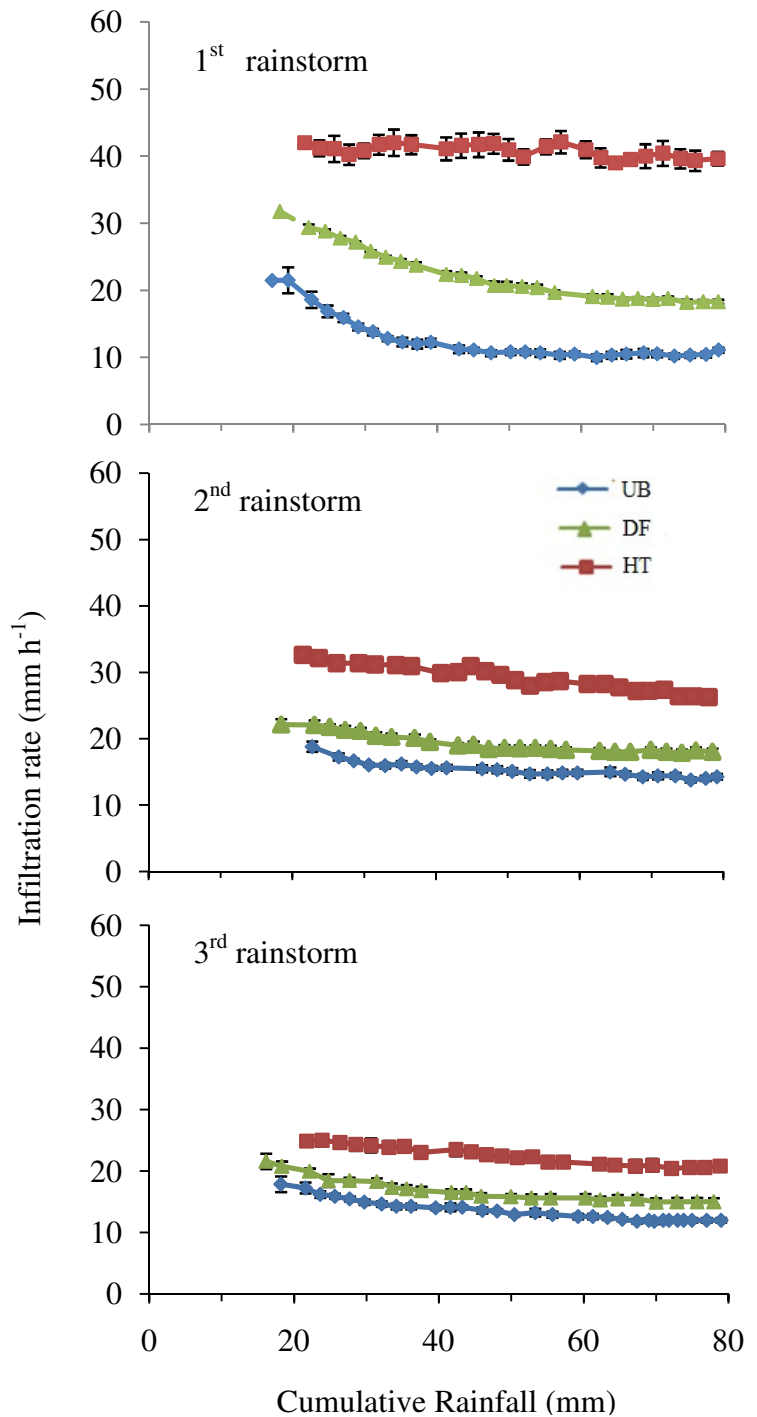

Fig. 3: *Infiltration rate as function of cumulative rainfall of three consecutive storms for UB, DF and HT soils. Bars indicate one standard error.*

values in the 3rd rainstorm, the UB and DF soils showed similar values (Fig. 4). Soil loss collected from the UB soil showed a similar pattern between one rainstorm to the next (Fig. 4), while in the two burnt soils increased in the 2nd rainstorm and 3rd rainstorms (Fig. 4).

The FIR values and total soil loss collected in the end of each rainstorm for the three soils are presented in Table 4. In all the rainstorms, FIR values were always in the following order: HT>DF>UB, but the differences among the three soils diminished in the 2nd and 3rd rainstorms. The FIR values of the UB soil increased significantly by ~34% between the 1st and 2nd rainstorms, and decreased significantly by 15% in the 3rd one. The FIR of the DF and HT soils decreased significantly in consecutive storms. However, the DF soil showed a less pronounced decline (i.e., a significant but very small decrease of 2% between the 1st and the 2nd storms, and 17% between the 2nd and 3rd) than the HT soil (i.e., a 33% decrease between the 1st and 2nd storms, and 23%
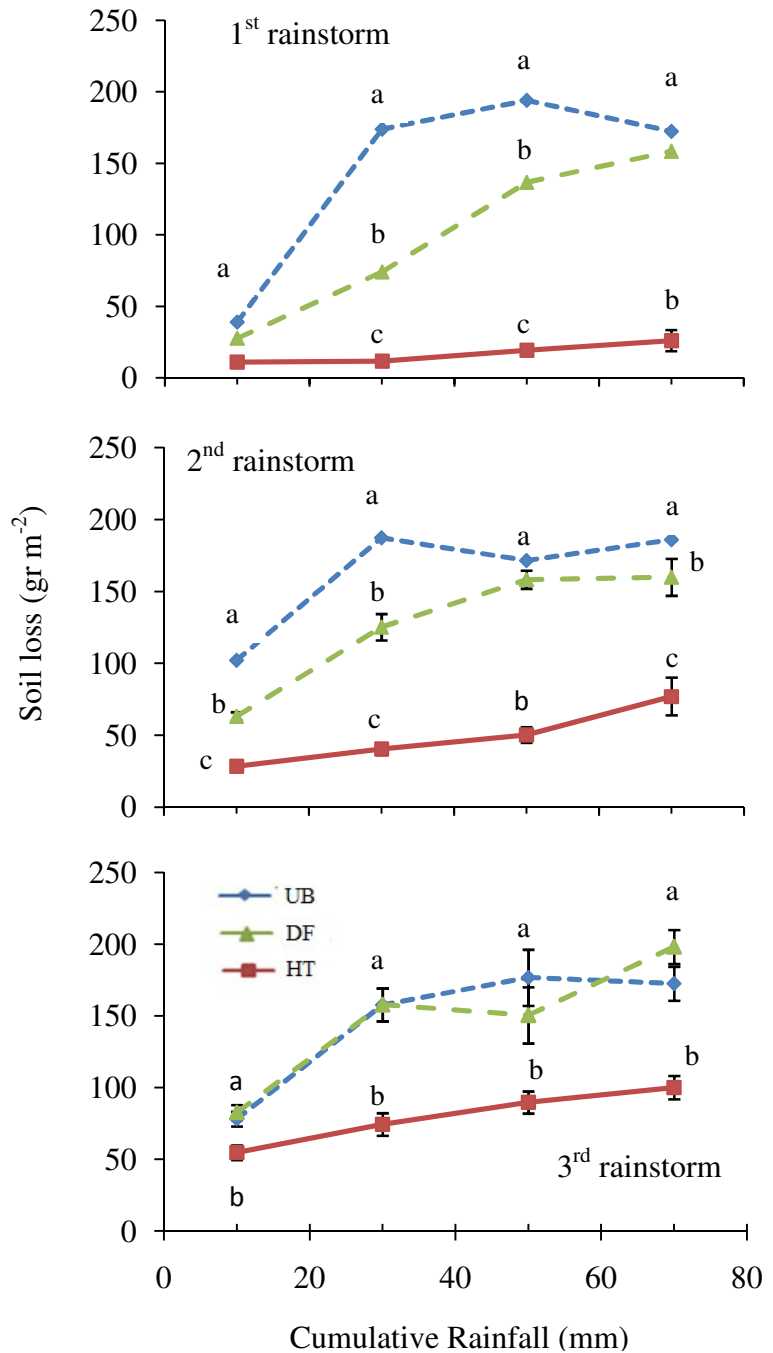

Fig. 4: Soil loss during three consecutive storms for UB, DF and HT soils as function of cumulative rainfall. Different lower case letters indicate significant differences between each treatment within a fraction ($\alpha \leq 0.05$). Bars indicate one standard error.

Table 4: *Final infiltration rate and total soil loss of UB, DF and HT soils collected after three consecutive storms. Different lowercase letter indicates significant differences between treatments within each storm. Different uppercase letters indicate significant differences between storms within each soil type.*

Soils	Final infiltration rate			Soil loss		
	1st Storm	2nd Storm	3rd Storm	1st Storm	2nd Storm	3rd Storm
	------------- mm h^{-1} -------------			------------- mm h^{-1} -------------		
UB	10.5 cC	14.1 cA	11.9 cB	580.0 aA	646.2 aA	585.0 aA
DF	18.4 bA	18.0 bB	15.0 bC	396.9 bC	506.0 bB	589.0 aA
HT	39.8 aA	26.6 aB	20.6 aC	68.0 cC	195.8 cB	318.8 bA

between the 2nd and 3rd). The increase of FIR in the UB soil between the 1st and 2nd storm could be a result of an increase in microbial activity during the drying period between storms, which releases extracellular polysaccharides that improve aggregation of soil particles and aggregate stability (Tisdal and Oades, 1983). However, in DF and HT soil, changes in soil properties due to fire or temperature could limit microbial development and therefore, no increase in FIR between the 1st and 2nd storms was observed. In general, final soil loss values were in following order in the 1st and 2nd rainstorms: UB>DF>HT. In the 3rd rainstorm, the UB and DF showed similar soil loss values, both significantly higher than that of the HT soil (Table 4). Both burnt soils showed a significant increase in soil loss from one rainstorm to the next. On the other hand, no significant differences were found between the soil loss values of the UB soil in all the rainstorms (Table 4).The significance of effects of soil treatment and rainstorm on FIR and soil loss is presented in Table 5. All fire treatment and rainstorms had a significant effect on FIR and soil loss values. Furthermore, a significant interaction ($p<0.0001$) between the soil and the rainstorms effect the FIR and soil loss.

4.1.1.5   The effects of fire on physic-chemical properties of the leachate/soil solution

As mentioned above, during the combustion of OM some organic molecules are

Table 5: *Significance of the different effects of treatment and rainstorm on final infiltration rate (FIR) and soil loss of Pale rendzina soil treated with different fire treatments.*

Variable	Source of variation	F value	P Value
FIR	Fire treatment	1595.662	<0.0001
	Rainstorm	394.078	<0.0001
	Fire treatment x Rainstorm	0.001	<0.0001
Soil loss	Fire treatment	205.820	<0.0001
	Rainstorm	12.358	<0.0001
	Fire treatment x Rainstorm	0.146	<0.0027

transformed into dissolvable organic substances (Gonzalez-Perez et al., 2004). Frenkel *et al,.* (1992) and Tarchitzky et al. (1999) found that the interaction of dissolved OM with clay particles increase dispersion of clay in suspension. The effect was attributed to the adsorption of negatively charged dissolved OM functional groups on the positively charged edges of the clay, which prevented edge to face flocculation of clay particles (Tarchitzky et al., 1999). Dissolved organic carbon in the leachate of the UB, DF and HT soils collected during three consecutive rainstorms as a function of cumulative rainfall is presented in Fig. 5. All soils showed a decrease in DOC as each rainstorm progressed. The HT soil consistently showed higher DOC than the other soils, whereas each of the UB and DF soils had similar values throughout all the rainstorms.

The EC and SAR changes in the leachate for UB, DF and HT soils collected during three consecutive rainstorms are shown in Figs. 6A and 6B, respectively, as functions of cumulative rainfall. The EC values of the leachate of all soils decreased as each rainstorm progressed, but with different rates for each soil and rainstorm (Fig. 6A). The EC of the HT soil decreased sharply from the 1st rainstorm to the 2nd, with no changes in trend from the 2nd to the 3rd rainstorm. However, these values were consistently and significantly higher from those observed in the other soils in all the rainstorms. The EC values of UB and DF soils did not differ

significantly between them except in the 1st fraction of the 1st and 2nd rainstorm, and showed a less pronounced decline than those of HT soil. The SAR values of all soils were low and decreased in all the storms as the rainfall progressed (Fig. 6B). During most of the 1st and 2nd storms, the SAR of the HT soil was significantly lower than that of the UB and DF soils, and these differences decreased as the number of rainstorms increased. In the last rainstorm, all the soils showed close SAR values, whereas the HT and DF soils showed lower SAR values than the UB soil (Fig. 6B).

4.1.2 Discussion

4.1.2.1 Effects of fire on soil physicochemical properties

It has been found that the fire treatments that were tested in this experiment has an effect on the texture of the Pale rendzina soil (Table 2): Exposure to prolonged, moderate temperatures heating treatment (i.e. HT soil) was found to decrease the clay fraction of the Pale rendzina soil by 35%, the sand fraction by 10% and to increase the silt fraction in 57% (Table 2). Ulery
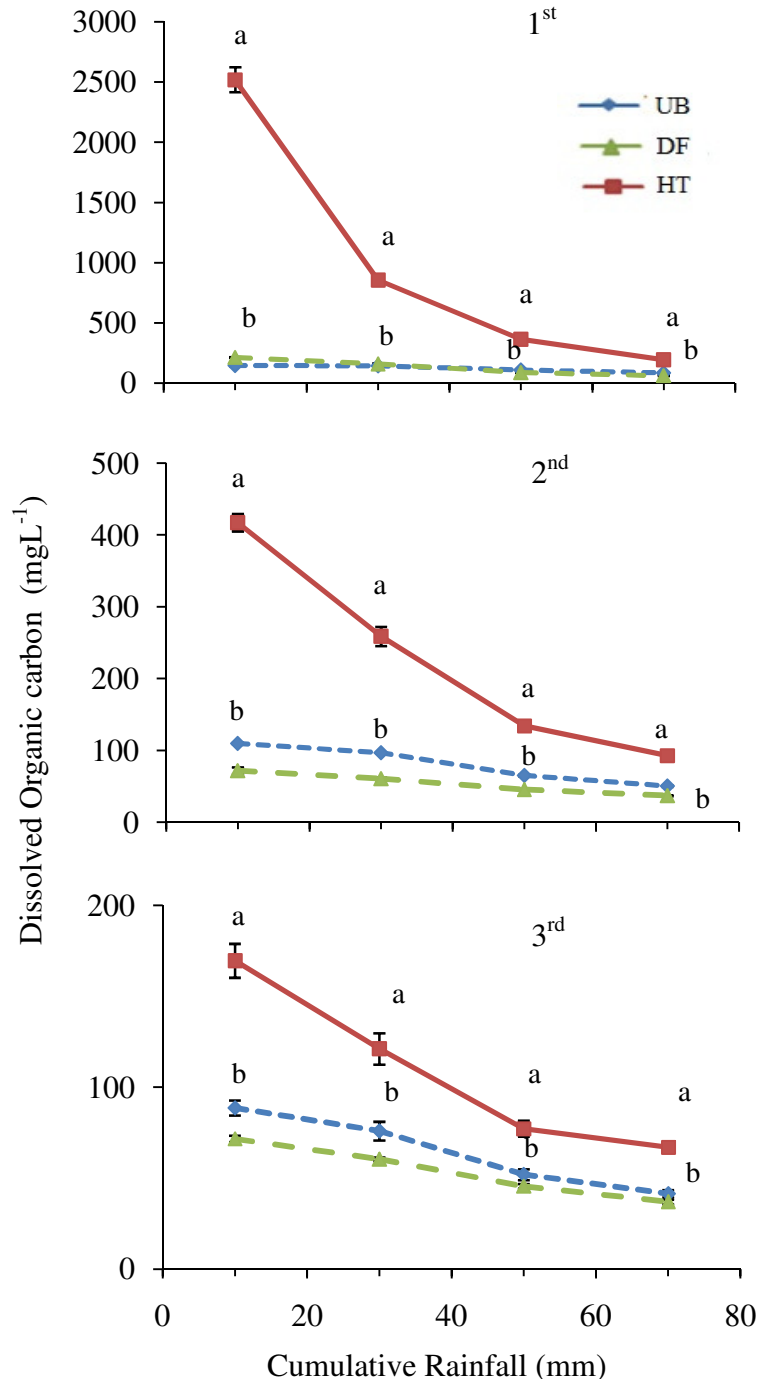

Fig. 5: Total dissolved organic carbon in the leachate of UB, DF and HT Pale rendzina soils collected during three consecutive storms as a function of cumulative rainfall. Bars indicate one standard error. Different lower case letters indicate significant difference between each soil type within one fraction ($\alpha \leq 0.05$).
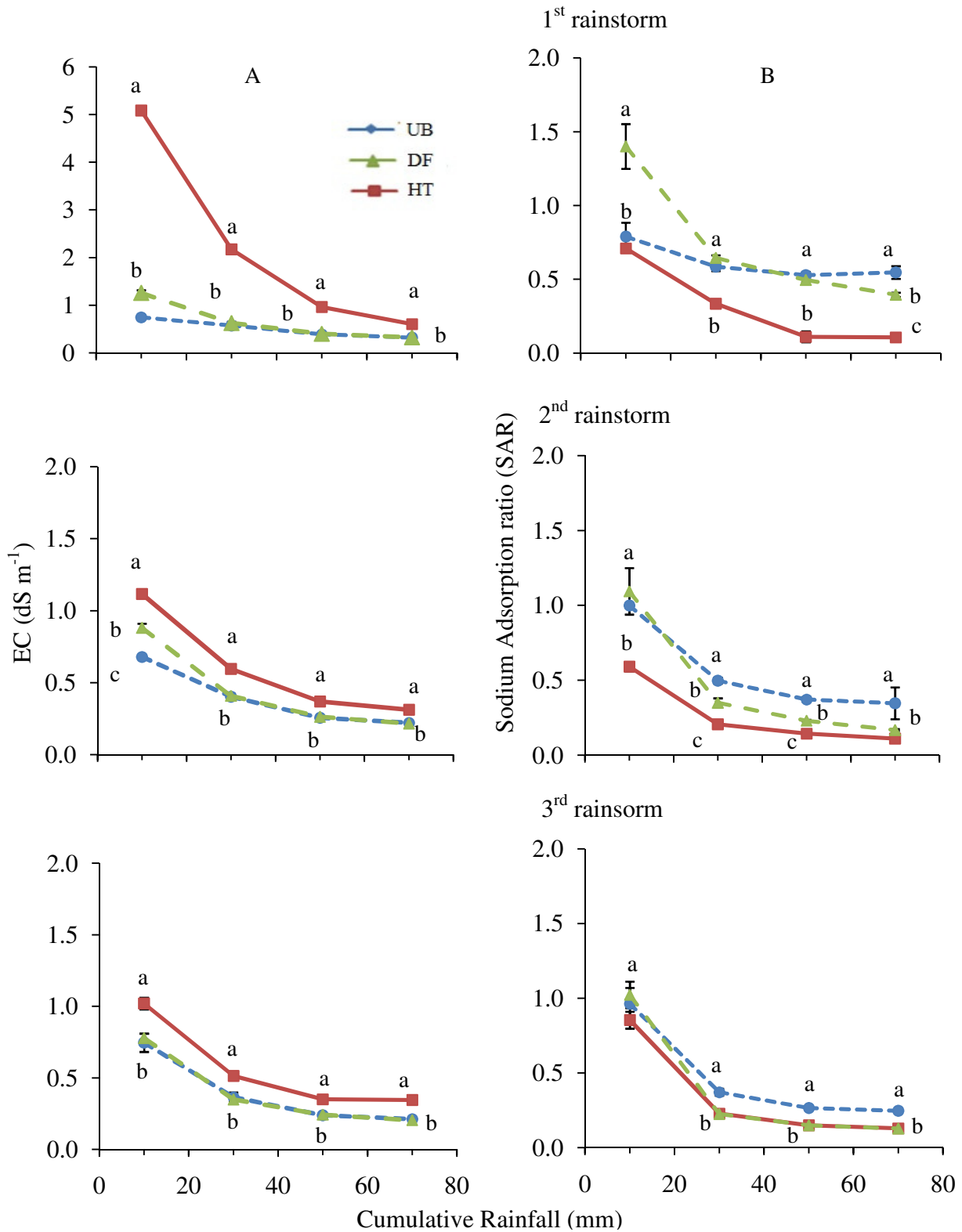
45

Fig. 6: *(A) Mean electrical conductivity (EC) and (B) mean sodium adsorption ratio (SAR) of the leachate of UB, DF and HT soils collected during three consecutive rainstorms as functions of cumulative rainfall. Bars indicate one standard error. Different lower case letters indicate significant difference between each soil type within one fraction ($\alpha \leq 0.05$).*

and Graham (1993) tested the effect of a severe wildfire conditions on the texture of five soils (four sandy loam soils and a loam soil) and found that the fire affected soils had coarser textural classes than the unburned ones. The authors found a significant decrease in the clay fraction in all of the soils, a significant increase in the silt fractions in one of the soils (Sandy loam), while the rest had an insignificant increase in the sand fraction (Ulery and Graham, 1993). Giovannini et al. (1988) found textural changes on silty clay and sandy loam soils subjected to temperatures over 220 ℃. The authors found that OM decreased dramatically in temperatures of 460 ℃ and over, clay fraction decreased and sand fraction increased, ending in a sandy textural class for both soils at 700 ℃ and 900 ℃. The authors postulated that heating promotes stronger cohesion in which iron and aluminum are involved in a process very similar to soil laterization (Giovannini et al., 1988). We postulate, that heating the soil to a temperature of 300 ℃ during 8 hours probably resulted in the fusion of clay particles into silt-size ones due to thermal transformation of iron and aluminum oxides (Giovannini et al., 1988; Terefe et al., 2008). Terefe et al. (2008) found a decrease in the clay fraction of some soils with varying pre-heating texture, following their exposure to 300 ℃. These authors postulated that aluminum oxides and hydroxides could be formed during clay decomposition, and they act as a cementing agent of clay particles resulting in strong and big tactoids of silt size. Thus, the significant increase of silt fraction may be a result of fusion of clay particles, and breakdown of sand sized into micro-aggregates with silt size due to a decrease in OM content (Table 2). From these results it could be assumed that the DF soil was exposed to temperatures below 300 ℃ and/or during shorter times than the HT soil, hence, keeping the original particle size distribution of the UB soil.

One of the soil properties affected by the increase of temperature is soil OM (Table 2). Changes in OM could lead to changes in soil physical properties, such as aggregate stability, since OM is essential for the formation and stabilization of soil aggregates, particularly in the upper horizons where most of it is concentrated (Tisdal and Oades, 1982; Chaney and Swift, 1984). Soil OM under prolonged exposure to moderate temperatures treatment (i.e. HT soil) was decreased by 30% (Table 2) probably as a result of combustion of OM when heating the soil at temperatures >200 ℃ (Giovannini *et al*,. 1988, 1990; DeBano et al., 1998). Under direct moderate fire conditions (i.e., DF soil) OM was increased by 65%. This increase might be due to the presence of ash mixed with soil particles that was not removed completely during soil sampling, and which contained partially decomposed OM (Gonzalez-Perez et al., 2004).

47

Gimeno-Garcia et al. (2000) and Gonzalez-Perez et al. (2004) also found an increase in OM content in soils after moderate fire conditions, and a decrease in OM content in soils after severe fire conditions. Although OM content of the HT soil decreased significantly compared to the UB soil, it showed higher dissolved organic carbon content than the other two soils (Table 2). Thus, it can be inferred that in the combustion of solid OM, insoluble organic compounds were transformed in more soluble ones. However, no significant differences were found between the dissolved organic carbon levels in the DF and in the UB soils (Table 2), and it can be attributed to one or more of the following reasons: (i) even though it caused a partial combustion of the duff layer and ground vegetation, the temperature at the ground level did not rise to levels that can cause a transformation of OM from insoluble to soluble form , and (ii) since the DF soil was sampled after several rainfall events, dissolved organic carbon from the ash that is originated from the combustion process of OM could have been washed by the rain and from upper soil layer.

The decrease in the CEC (and the parallel increase in ESP) in the HT soil can be attributed to the decrease in the clay fraction and OM content, which reduces the number of exchangeable sites in soil colloids (Giovannini et al., 1990; Neary et al., 2005). Similarly, the increase of CEC in DF was probably a result of the increase in OM content in this soil that could be originated from ash particles mixed with the soil while sampling it. The increase in the calcium carbonate content in the HT soil (Table 2) can be a consequence of the formation of calcite in the ash, and from carbonates originated during the combustion of OM (Ludwig et al., 1998; Ubeda et al., 2009). or a result of their precipitation from bi-carbonate created during the combustion of soil OM, and Ca^{2+} cations released from the exchange sites as they were replaced by NH_4 ions (Neary et al., 2005). However, this issue has to be further studied. Wildfire was found to increase pH in Mediterranean calcareous soils (Kutiel and Naveh, 1987; Hernandez et al., 1997), mainly by input of ash, the release of cations during the combustion of OM, and the denaturation of organic acids (Certini, 2005; Terefe et al., 2008). However, Certini (2005) notes that fire induced changes in pH are negligible in soils buffered by carbonates. In the present study, none of the fire treatments caused a significant change in pH (Table 2). These results are similar to those of Kutiel and Inbar (1993), who did not find changes in pH of Pale rendzina soils exposed to fire in an unknown intensity.

The electrical conductivity (EC) represents the electrolyte concentration of the soil solution and can affect the tendency of the soil particles to flocculate or disperse (Quirk and Schofield, 1955; Agassi et al., 1981; Shainberg and Letey, 1984; Ben-Hur. 2008). In this study, a significant increase in the EC was observed in the HT soil as compared to the UB and DF soils, probably as a result of the intense heat in the muffle that can cause a thermal transformation of OM and clay particles which can release soluble ions to the soil solution (Giovannini et al., 1990). An increase in EC following fire was also found by other studies (e.g., Kutiel and Naveh, 1987; Kutiel and Inbar, 1993; Kutiel et al., 1995; Hernandez et al., 1997; Iglesias et al., 1997; Pardini et al., 2004; Terefe et al., 2008). Kutiel and Naveh (1987) found an increase in soil EC following a high severity wildfire and ascribed it to the release of soluble cations from the ash originated from the combustion of OM. However, the values of EC rapidly decreased to initial levels due to washing of the ions with percolating rain water. The latter phenomenon can explain the lower EC values in the DF soil (Table 2), since this soil was sampled after a few precipitation events that may have washed these fire-induced soluble ions from the upper soil profile.

At a given ESP value, clay dispersion increases as EC of the soil solution decreases (Shainberg and Letey, 1984). Similarly, low SAR values for a soil indicate existence of higher concentration divalent cations in the soil solution such as Ca^{2+} and Mg^{2+} as compared to Na^+ (Eq., [2]; Shainberg and Letey, 1984). Hence, the lower the SAR, the more stable the soil is to clay dispersion, because it keeps the electrolyte concentration of the soil solution above the flocculation value of the clay (Agassi et al., 1981; Shainberg and Letey, 1984; Ben-Hur, et al., 2008). In general, all the soils showed low and similar SAR values, which can result in a low tendency for clay particles to disperse (Shainberg and Letey, 1984). However, the burnt soils had significantly lower SAR compared to the UB soil, probably as a result of a large release of divalent soluble cations from thermal transformation of soil OM and clay particles (Giovannini et al., 1990) and from the ash (Kutiel and Naveh, 1987).

4.1.2.2   Effects of fire on IR/runoff/soil loss relationships

In general, in all the soils and all the rainstorms, the increase in soil loss (Fig. 4) is a result of two main processes: (i) the destruction of soil structure by raindrop impact increases detachment of soil particles; and (ii) the formation of a seal on the soil surface which cause a decrease in the IR of the soil during the rainstorms (Fig. 4) and results in an increase in runoff amounts (data not shown) and its transport capacity (Agassi et al., 1981; Morin et al., 1981).

Throughout the rainstorms, in all fire treatments (Fig. 6), there is a decrease in the EC values due to washing of electrolytes by the percolating water This decrease in the EC may result in an increase in clay dispersion (Quirk and Schofield, 1955, Agassi et al., 1981; Shainber and Letey, 1984), and can explain the decrease in infiltration rate and the increase in soil loss within each rainstorm. Changes in soil properties, especially OM, could alter soil hydraulic properties, among them infiltration rate and soil loss (DeBano et al., 1998; Prosser and Williams, 1998; Robichaud and Brown, 1999; Moody and Martin, 2001; Benavides-Solorio and MacDonald, 2005). As showed before, the fire effect on the soil, both in the DF and HT soils, produced changes in their properties (Table 2). We postulate that these changes resulted in higher IR and lower soil loss in all three rainstorms in the HT soil and in the first two rainstorms in the DF soil, compared to the UB soil (Fig. 3, Fig. 4 and Table 4).

We suggest three possible causes for the differences in IR and soil loss among treatments: (i) soil water repellency caused or destructed by fire or temperature (DeBano, 1981, 2000; Robichaud, 2000; Shakesby et al., 2000); (ii) changes in the solid phase of the soil that can affect its aggregate stability and its ability to withstand external physical forces such as raindrop impact and fast wetting, and (iii) physicochemical changes in the soil solution that can affect the dispersion of clay particles and cause soil swelling, which are directly related to the formation of a structural seal on the soil surface (Quirk and Schofield, 1955; Agassi et al., 1981; Morin et al., 1981; Shainberg and Letey, 1984; Ben-Hur, 2008).

Since no significant water repellency was found (see chapter 3.2.3), in all the soils and within each rainstorm, it can be concluded that the decrease of IR is a result of aggregate disintegration, clay dispersion and subsequent seal formation at the soil surface (Morin et al., 1981; Assouline, 2004). Furthermore, the lack of differences in the water repellency between the three soils (data not shown) indicates that it cannot be the cause of the differences between their IR, runoff and soil loss values (Fig. 3; Fig 4 and Table 4).

Aggregate stability tests were conducted in order to determine the aggregate's ability to withstand the slaking forces when wetted fast (Ben-Hur et al., 2009). From the lowest OM content of the HT soil compared with the other two soils (Table 2), it could be expected that its aggregates would be weaker and less resistant to slaking forces during fast wetting (Lado et al., 2004b). However, the HT soil showed the lowest SLV as compared with the UB and DF soils, suggesting that aggregates from this soil are less sensitive to slaking (Fig 2). Varela et al. (2010)

found that when heating some forest soils to 220 ℃ their aggregate stability increased, but, when they heated these soils to higher temperatures, the aggregate stability dramatically decreased. The authors suggested that heating the soils to 220 ℃ induced the formation of a water repellent layer surrounding soil aggregates that stabilized them, but when the soils were heated to temperatures higher than 288℃, the protective repellent layer disappeared, hence the reduction in the aggregate stability (Varela et al., 2010). In the present study, however, no water repellency was observed in the soils (data not shown), and therefore it can be assumed that changes in aggregate stability are not a result of the presence of a water repellent layer covering the aggregates. Similar increase in aggregate stability after heating the soils was found by Giovannini et al. (1988), who hypothesized that it might be a result of clay fusion into sand sized particles after exposure to high temperatures, as postulated before, which can explain the reduction in clay fraction in the HT soil (Table 2). Since no differences were found between the SLV of the UB and DF soils (Fig. 2), it can be assumed that the direct wildfire did not have any noticeable effect on the stability of the aggregates. In DF soil, the increase of temperature in the soil did not cause a change in texture and therefore it is reasonable to assume that no clay fusion took place due to the low temperatures reached during heating (Table 2). Moreover, the increase of OM content in the DF soil (Table 2) compared to UB soil do not result in a higher resistance to slaking forces, which is in accordance with the external origin of this increase of OM content in the DF soil, since external OM does not contribute to aggregate stability (Fox et al., 2007).

The results of the dispersion test indicate that the HT soil was less dispersive than the UB and DF soils, showing higher structural stability (Table 3). This difference can be a result of the highest EC and lowest SAR of the soil solution of the HT soil (Table 2 and 3), probably originated by the thermal transformation of OM and clay particles that released divalent cations and soluble molecules, keeping the EC above the flocculation value of the clay (Quirk and Schofield, 1955, Agassi et al., 1981; Shainberg and Letey, 1984; Ben-Hur, et al., 2008). In the rainfall simulator, the EC of the soil solution of the HT soil was the highest in all three rainstorms (Fig. 6A), and its SAR was lower in the first two rainstorms than the other two soils (Fig. 6B). These differences, added to the higher resistance to slaking of this soil, may be the reason for the higher IR and lower soil loss in of the HT soil in all the rainstorms. Less clay dispersion results in a seal on the soil surface that is less dense, and with higher hydraulic conductivity compared to the UB and MS soils. Furthermore, DOC-induced clay dispersion did

not play a major role on the IR of the HT soil since higher DOC concentrations were found in the soil chemical properties and during all three rainstorms (Table 2, Figure 5), compared to the other two soils (Frenkel *et al*,. 1992; Tarchitzky et al., 1999). The differences in EC and SAR values between the soils (Fig. 6) diminished between rainstorms, resulting in less difference in IR values and soil loss between soils (Fig 3 and Table 4).

Unlike the HT soil, the DF soil did not show differences in EC and SAR (Table 2 and Fig. 6) probably because the low-moderate wildfire was not intense enough to release cations and soluble molecules to the soil solution, as was found by others (e.g., Kutiel and Naveh, 1987; Kutiel and Inbar, 1993; Kutiel et al., 1995; Hernandez et al., 1997; Iglesias et al., 1997; Pardini et al., 2004; Terefe et al., 2008). Since the DF soil was sampled after the onset of the rainy season, another explanation to the aforementioned result can be that cations and soluble molecules from the DF soil were leached by rainwater (Kutiel and Naveh, 1987) prior to the sampling time. The lack of significant difference between the EC and SAR in the leachate of the DF and UB soils (Fig. 6) can explain their similar dispersion values (Table 3). Nevertheless, the DF soil showed lower SAR, both in the soil solution composition test (Table 2) and in the dispersion test itself (Table 3) compared to the UB soil. Thus it can be assumed that the aforementioned differences in SAR between these two soils are not enough to show any effect on dispersion, or that the method used to measure the dispersion value was not sensitive enough to show differences in dispersion values in those conditions (Table 3). As postulated before, the direct moderate wildfire resulted in higher IR and lower soil loss in the first two rainstorms than those observed in the UB soil (Fig. 3, Fig. 4, Table 4), even though the differences in soil structure between the DF and UB soils could not be explained by differences in slaking and dispersion (Fig. 2 and Table 3, respectively). These differences between the UB and DF soils in IR, runoff and soil loss in the 1st and 2nd rainstorms needs to be further investigated.

The above mentioned results are relevant to understand the processes that lead to runoff and soil loss after a forest fire. The fire treatments in this chapter represent three possible situations that could exist in a patched landscape after a wildfire. Some patches could not be affected by the fire due to large distance from combustible fuel source; some can be affected by a direct low-moderate fire, while other patches could experience prolonged soil heating with no direct fire contact such as soil impacted by radiated heat from a nearby smoldering of fuels. The fire treatments caused an increase in infiltration and a soil loss reduction by making changes in

the chemical and physicochemical properties of the Pale rendzina soil. Nevertheless, consecutive rainstorms decreased the fires' stabilizing effects. According to our results, no additional protection provided to the surface of a bare soil after a forest fire, could result in an increase in soil erosion risk.

4.2   Part II: Use of PAM to reduce soil loss – Rainfall simulator study

4.2.1   Results

4.2.1.1   Effect of PAM on IR, runoff and soil loss of Pale rendzina and BSL soil

In the previous chapter, we observed the effect of different fire severities on soil properties, IR, runoff and soil loss in a Pale rendzina soil. We found that fire was changing the chemical and physical properties of the soil and physicochemical properties in the soil solution. However, the absence of a soil cover, which is usually lost by combustion during the fire, can result in a net increase of overland flow and soil erosion, since the bare soil is exposed to direct impact of the raindrops. Therefore, the implementation of protection measures to reduce soil loss after fire is a crucial step to avoid losing the soil and to enhance vegetation recovery.

Infiltration rate and soil loss of UB, DF and HT soils treated with 0 (control treatment) and 50 kg ha^{-1} granular PAM (PAM treatment), during three consecutive storms as functions of cumulative rainfall are presented in Figs. 7, 8 and 9, respectively. Likewise, FIR values, runoff yield and total soil loss for each soil, treatment and storm are presented in Table 6. All the soils showed the same trend in relation to IR: Applying 50 kg ha^{-1} PAM decreased IR during the 1st rainstorm in all the soils, resulting in significantly lower FIR and higher runoff yield than control soils (Table 6). However, the differences disappeared after the first drying cycle, and in the 2nd and 3rd rainstorms, PAM treated soils showed similar or higher IR than the untreated ones (with the corresponding inverse response of the runoff).

On the other hand, the application of PAM to the soil surface reduced soil loss in UB and DF soils throughout the three rainstorms (Figs. 7 and 8 and Table 6). Despite the higher runoff amount observed during the 1st rainstorm in these soils, soil loss was lower in the PAM-treated than in the control soil. The differences were higher in UB soil, where total soil loss decreased by 64% (Fig. 7B and Table 6), than in DF soil, where this reduction was 44% (Fig. 8B and Table 6). The HT soil showed a slightly different behavior: soil loss increased with the PAM treatment in the 1st rainstorm (Fig. 9B and Table 6).

Results of a multifactor analysis of variance are presented in Table 7. The results indicated that each main variable by itself (i.e., fire treatment, rainstorm, and PAM treatment) significantly affected FIR, runoff and soil loss (Table 7). The interactions: fire treatment x
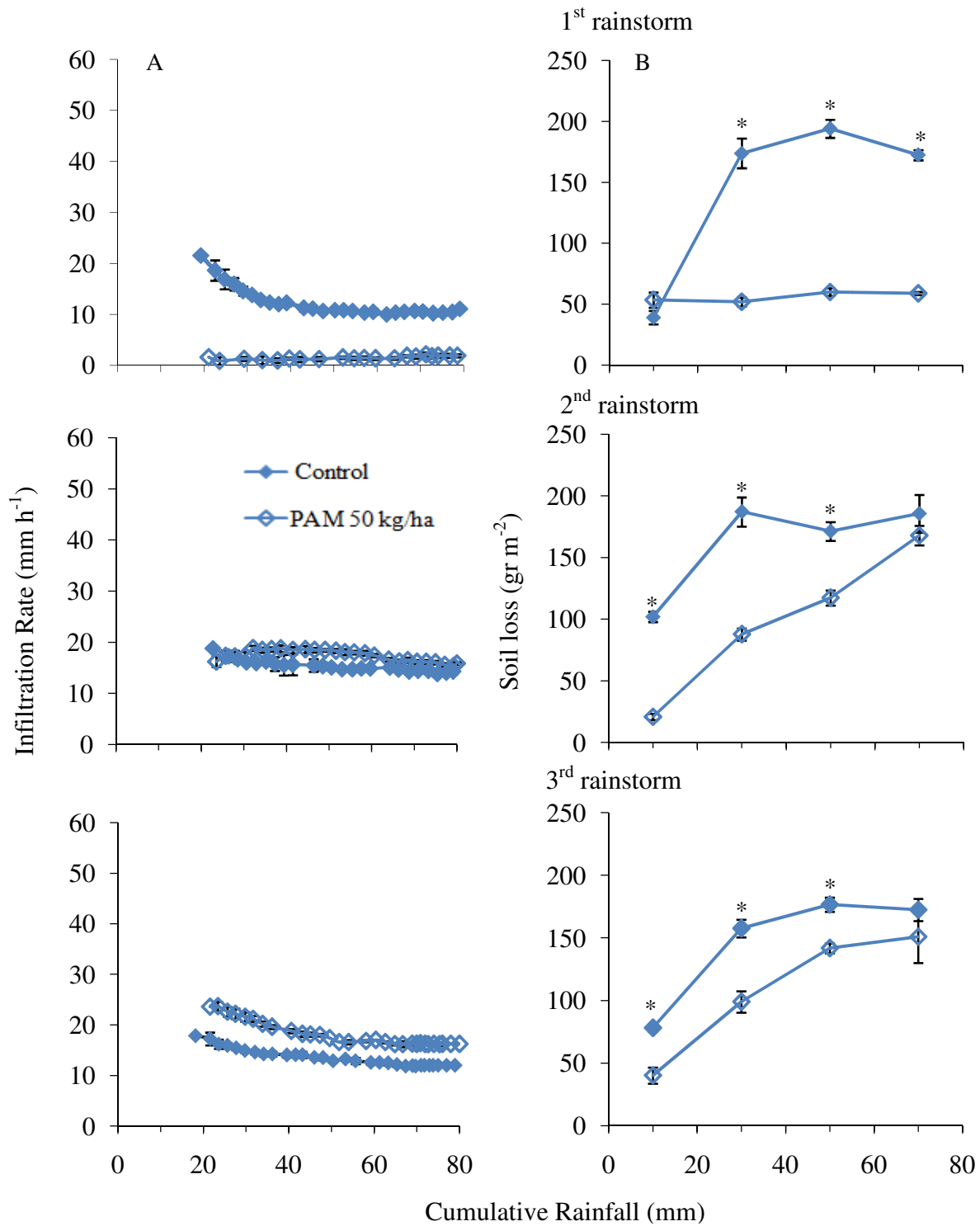
Figure 7*: (A) Infiltration rate and (B) soil loss of UB soil during three consecutive rainstorms after control (0 kg ha⁻¹ PAM) and 50 kg PAM ha⁻¹ treatments as function of cumulative rainfall. For soil loss, an asterisk indicates significant differences between treatments within each fraction (α≤ 0.05). Bars indicate one standard error.*
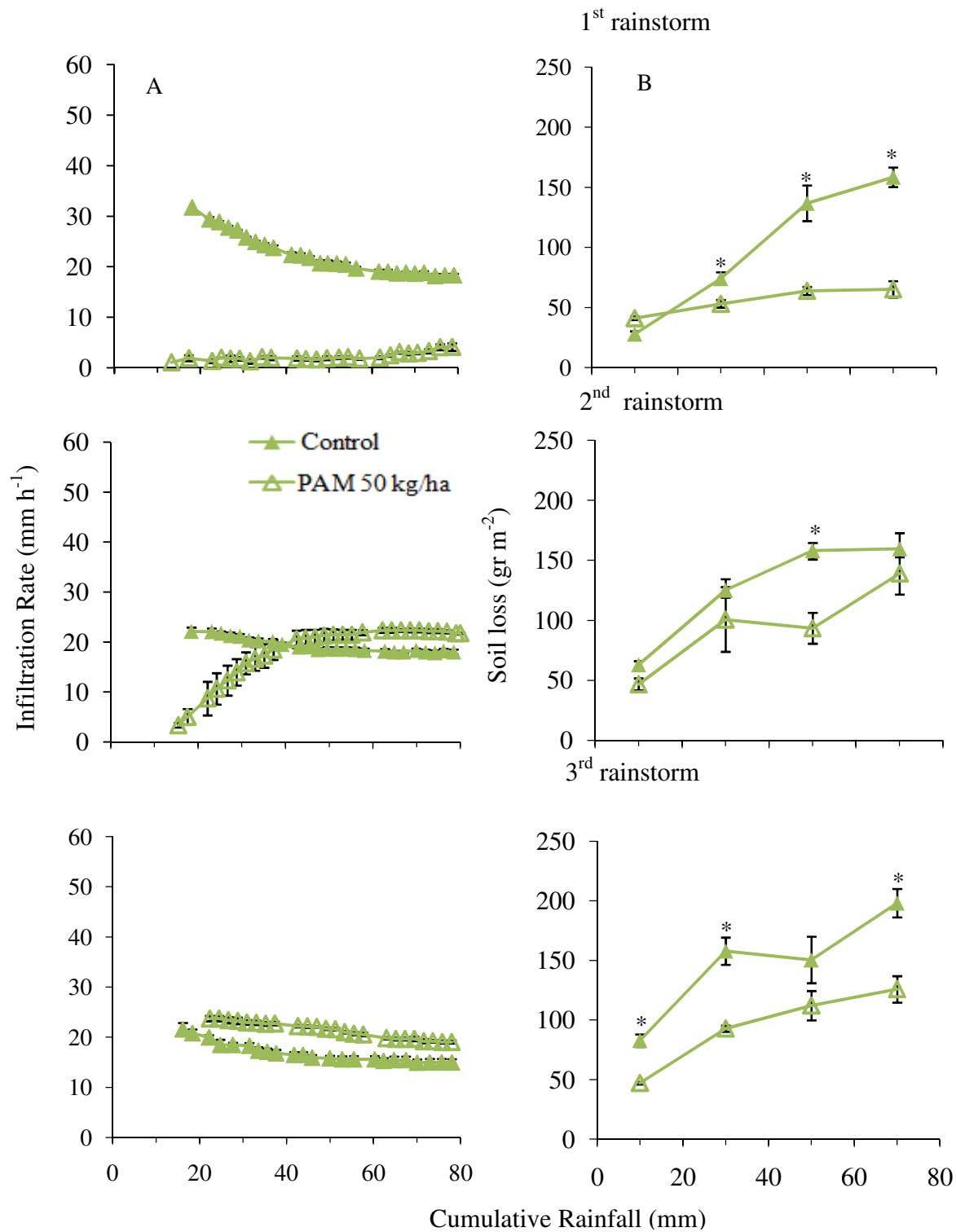
Figure 8*: (A) Infiltration rate and (B) soil loss of DF soil during three consecutive rainstorms after control (0 kg ha⁻¹ PAM) and 50 kg PAM ha⁻¹treatments as function of cumulative rainfall. For soil loss, an asterisk indicates significant differences between treatments within each fraction (α≤ 0.05). Bars indicate one standard error.*
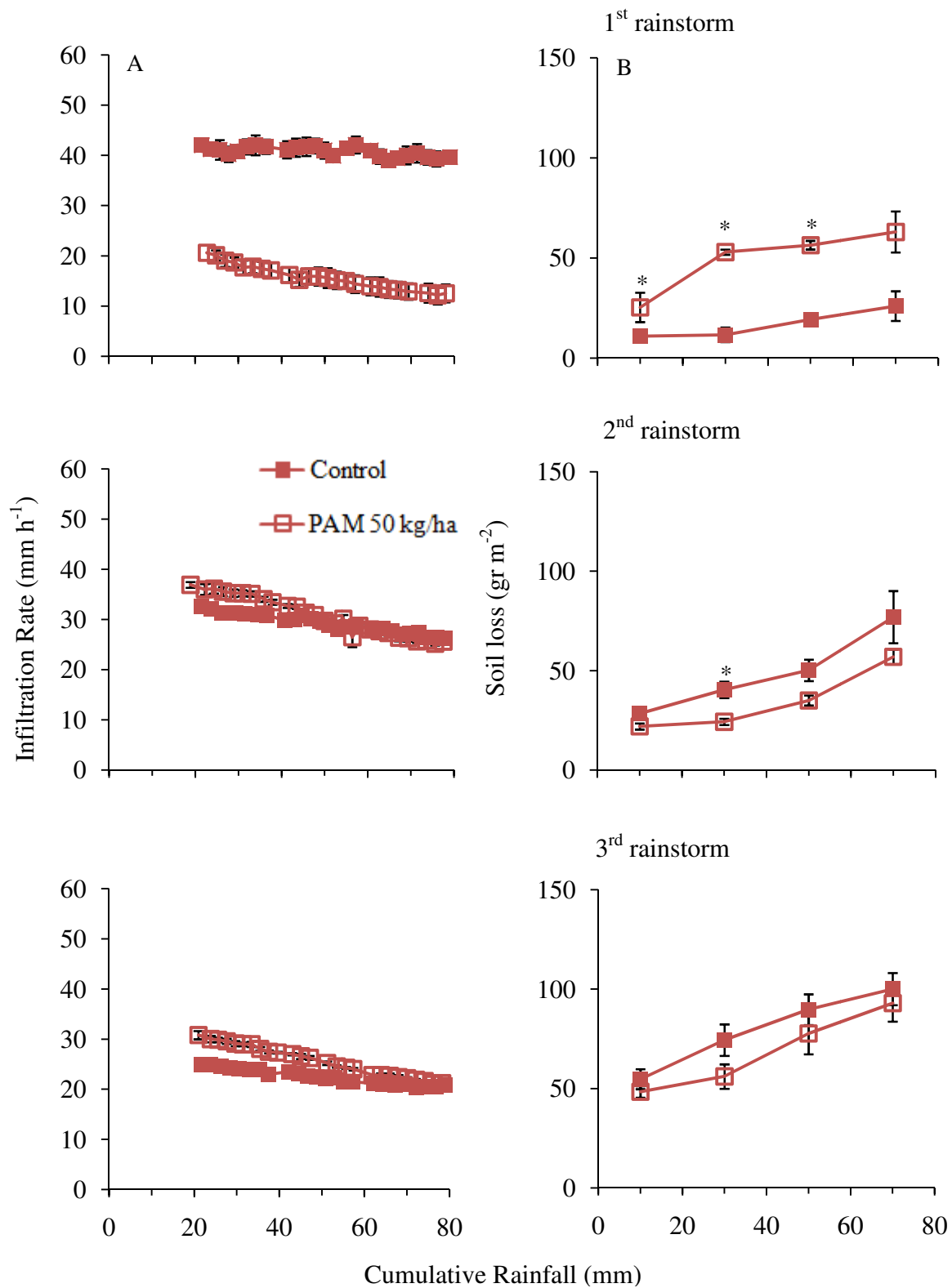
Figure 9: (A) Infiltration rate and (B) soil loss of HT soil during three consecutive rainstorms after control (0 kg ha^{-1} PAM) and 50 kg PAM ha^{-1} treatments as function of cumulative rainfall. For soil loss, an asterisk indicates significant differences between treatments within each fraction (α≤ 0.05). Bars indicate one standard error.

57

Table 6: *Final infiltration rate (FIR), total runoff and total soil loss, for UB, DF and HT untreated (control) and treated (PAM) Pale rendzina soils and heat treated sandy loam soil from Barbanza, Galicia, Spain (BSL), during three consecutive rainstorms. Different lowercase letters indicate significant differences between PAM and control for each soil within each storm. Difference uppercase letters indicate significant differences between rainstorms within each treatment.*

Variable	Soils	1st storm		2nd storm		3rd storm	
		Control	PAM	Control	PAM	Control	PAM
FIR		------------------------------- mm h^{-1} --------------------------------					
	UB	10.5 aA	1.9 bC	14.1 bC	15.7 aB	11.9 bB	16.3 aA
	DF	18.4 aA	3.6 bC	18.0 bB	21.9 aA	15.0 bC	19.1 aB
	HT	39.8 aA	12.7 bC	26.6 aB	25.6 bA	20.6 bC	21.5 aB
	BSL	34.2 aA	na	29.0 bB	32.7 aA	25.0 bC	27.6 aB
Runoff		----------------------------------- mm -----------------------------------					
	UB	34.0 bA	51.4 aA	31.0 aB	29.7 aB	32.0 aAB	24.9 bC
	DF	18.8 bC	50.6 aA	23.6 aB	25.4 aB	27.9 aA	23.1 bB
	HT	3.7 bC	31.3 aA	17.5 aB	10.7 bC	23.0 aA	19.8 aB
	BSL	5.6 bC	36.0 aA	10.5 aB	6.5 aB	15.7 aA	12.0 bB
Soil loss		----------------------------------- g m^{-2} -----------------------------------					
	UB	580.0 aA	209.3 bB	646.2 aA	394.0 bA	585.0 aA	432.1bA
	DF	396.9 aC	222.8 bB	505.9 aB	379.9 bA	589.0 aA	377.5 bA
	HT	68.0 bC	197.8 aB	195.8 aB	138.2 bC	318.8 aA	275.0 aA
	BSL	364.2 aB	226.3bB	629.4 aAB	413.5 aAB	905.3 aA	738.6 aA

Table 7: *Significance of effects of fire treatment, rainstorm and polymer treatment on final infiltration rate (FIR), runoff and soil loss on Pale rendzina fire treatments.*

Variable	Source of variation	F value	P value
FIR	Fire treatment	1145.307	<0.0001
	Rainstorm	194.673	<0.0001
	Polymer treatment	171.137	<0.0001
	Fire treatment x rainstorm	0.001	<0.0001
	Fire treatment x polymer treatment	132.727	<0.0001
	Rainstorm x polymer treatment	705.247	<0.0001
	Fire treatment x rainstorm x polymer treatment	134.834	<0.0001
Runoff	Fire treatment	33.710	<0.0001
	Rainstorm	13.124	<0.0001
	Polymer treatment	7.043	<0.0001
	Fire treatment x rainstorm	0.094	<0.0001
	Fire treatment x polymer treatment	1.395	<0.0001
	Rainstorm x polymer treatment	45.002	<0.0001
	Fire treatment x rainstorm x polymer treatment	0.180	<0.0001
Soil loss	Fire treatment	128.900	<0.0001
	Rainstorm	11.577	<0.0001
	Polymer treatment	45.262	<0.0001
	Fire treatment x rainstorm	0.288	<0.0001
	Fire treatment x polymer treatment	28.670	<0.0001
	Rainstorm x polymer treatment	0.009	<0.9280
	Fire treatment x rainstorm x polymer treatment	0.107	<0.0001

rainstorm, fire regime x polymer treatment, fire regime x rainstorm x polymer treatment significantly affected FIR, runoff and soil loss.

In order to test if the effects of PAM on runoff and soil loss observed in the Pale rendzina exist in other soils, the same simulated rainfall experiments were reproduced using a heat treated non calcareous sandy loam soil (BSL) from Barbanza, Galicia (NW Spain). Some properties of this soil are presented in Table 1. This soil was chosen since it represents very contrasting conditions with respect to the Pale rendzina. The PAM treatments were applications of 0 and 50 kg ha^{-1}. Infiltration rate and soil loss of BSL burnt soil after control (0 kg ha^{-1} PAM), and PAM (50 kg ha^{-1} PAM) treatments as functions of cumulative rainfall during three consecutive rainstorms are presented in Fig. 10. FIR, runoff and soil loss are presented in Table 6. During the 1st rainstorm, untreated soil showed relatively high IR, with only a slight reduction during that resulted in high FIR and low runoff (Table 6). The high IR is probably a result of the sandy loam texture, which results in a pore system dominated by large pores, and a clay fraction dominated by kaolinite. Furthermore, the heating of this soil might have caused an increase in aggregate stability due to fusion of clay particles (Giovannini et al., 1988) as was found in the HT rendzina soil, however, soil texture measurements after the heat treatment were not conducted. Moreover, the natural WR existent in this soil was destroyed during the heating, since water repellant substances were found to be combusted in 288°C (DeBano et al., 1998). In this rainstorm, the PAM treatment resulted in low IR at the beginning, but it increased constantly and the final IR after 80mm of cumulative rainfall was close to the one of the non treated soil (Fig. 10). FIR value could not be calculated for this soil since IR did not reach a steady state at the end of the 80 mm rainstorm (Table 6). Soil loss in both PAM and control treatments showed similar increase in the first 40mm, but while the soil loss in the PAM treatment declined after 40 mm of rain, in the control increased constantly second half of the storm (Fig. 10B). Total soil loss was significantly lower in the PAM treatment than in the control, while total runoff expressed an opposite trend (Table 6), in a similar trend that the one observed in UB and DF soils.

In consecutive rainstorms, after the first drying periods, IR of the PAM-treated BSL soil increased and reached higher IR values than the ones from the control one (Fig. 10A; Table 6). Consequently, lower total runoff amounts were obtained after the PAM treatment, although only in the last rainstorm they were statistically significant (Table 6). Soil loss measurements were

lower in the PAM-treated soil than in the control in all the rainstorms (Fig. 10B; Table 6), although in the 2nd and 3rd rainstorms the differences were not statistically significant (Table 6).

4.2.1.2   Effect of PAM on the viscosity and quality of the runoff

Relative viscosity measurements of the runoff collected during the 1st rainstorm in the control and PAM-treated for UB, DF, HT and BSL soils are presented in Fig. 11 as functions of cumulative rainfall. Relative viscosity of the leachate in all PAM-treated soils was not significantly different than the one of the untreated controls, and therefore this data is not presented. The absence of differences in the leachate indicates that PAM molecules that penetrated the soil were adsorbed to the soil particles during the percolating process, and their desorption was negligible (Nadler et al., 1992). However, the relative viscosity values of the surface runoff of the PAM-treated soils were significantly higher than their untreated control counterparts at the beginning of the 1st rainstorm, and were in the following order: BSL>UB=DF>HT (Fig. 11). Relative viscosity values decreased as the storm progressed to reach similar values in both treatments towards the end in all the soils (Fig. 11). Relative viscosity of all PAM-treated soils did not differ from the controls in the 2nd and 3rd rainstorms (data not shown).

The EC values measured in the runoff of the 1st rainstorm in the untreated UB, DF, HT and BSL soils are presented in Fig. 12A and represent the electrolyte concentration throughout the 1st rainstorm. In the Pale rendzina soil, The EC values were in the following order: HT>DF>UB. In the first 20 mm of the rainstorm, the EC value of the BSL soil was similar to that of the DF soil, while in the last 60 mm it descended to values similar to that of the UB soil (Fig. 12A). The pH measured in the runoff as a function of cumulative rainfall of the untreated Pale rendzina and BSL soils in the 1st rainstorm are presented in Fig. 12B. The pH of the Pale rendzina soils was higher than the pH of the BSL soil. The reason for the difference in pH is found in both the parent material and the climate in which these two soils were originated. Pale rendzina is originated from calcareous sedimentary rocks, rich in Ca^{2+} and Mg^{2+}, and in Mediterranean areas with relative low precipitation, therefore its pH is usually neutral to basic. The BSL soil is originated from granitic rocks poor in bases and rich in acidic cations. Moreover, the high precipitation in this area (it can reach over 2000mm yr^{-1}) depletes this soil from basic cations and hence the pH is acidic.
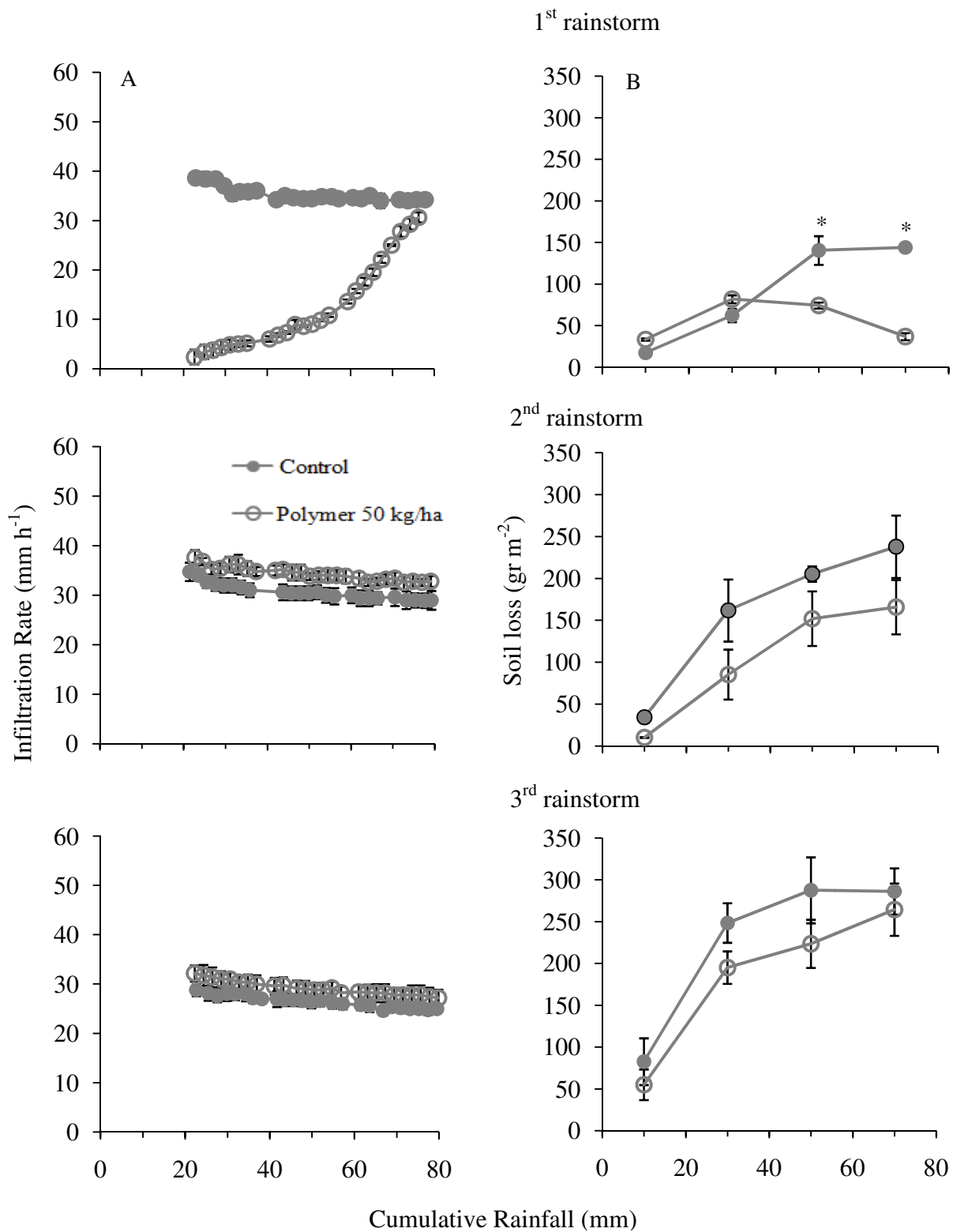
61

1st rainstorm

2nd rainstorm

3rd rainstorm

Figure 10: (A) Infiltration rate and (B) soil loss of the BSL soil during three consecutive rainstorms after control (0 kg ha^{-1} PAM) and 50 kg PAM ha^{-1} treatments as function of cumulative rainfall. For soil loss, an asterisk indicates significant differences between treatments within each fraction ($\alpha \leq 0.05$). Bars indicate one standard error.
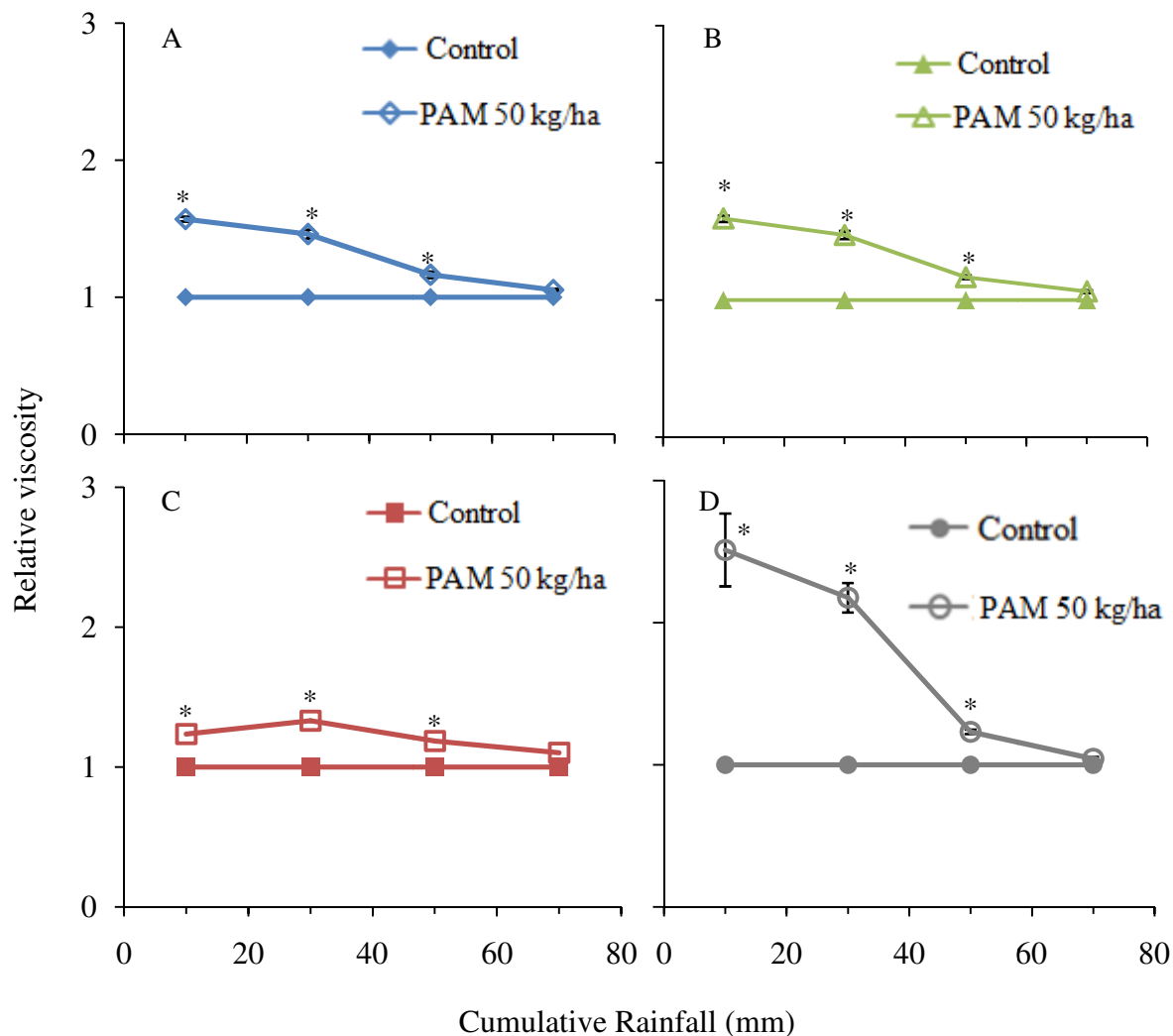
Figure 11: *Relative viscosity of the runoff collected from (A) UB, (B) DF, (C) HT Pale rendzina fire treatments and (D) BSL soil, treated with 0 kg ha⁻¹ (control) and 50 kg ha⁻¹ PAM, during the first storm in the rainfall simulator experiment. Asterisks indicate significant difference between treatments within each fraction ($\alpha \leq 0.05$). Bars indicate one standard error.*

4.2.1.3 Effect of PAM on aggregate stability

PAM application could affect IR through an increase of aggregate stability, which results in the formation of a less dense seal and therefore higher IR under raindrop impact (Shainberg et al.,. 1992; Levy and Ben-Hur, 1998; Aese et al., 1998; Levy and Miller, 1999; Ben-Hur, 2006; Mamedov et al., 2007; Sojka et al., 2007). Slaking and dispersion tests were made using the Ben-Hur et al. (2009) method, in order to examine the effect of PAM on soil structure. The results of the slaking experiment for untreated and treated (0 and 50 kg ha-1 PAM, respectively) DF soil

are presented in Fig. 13. Mean weight diameter (MWD) of fast wetting was higher in the PAM treated soil than the untreated control (Fig. 13). As a result, the SLV of the PAM treated soil was lower than the control soil, indicating higher aggregate stability (Fig. 13). In fact, SLV value indicates that almost no slaking occurred when PAM was applied, since there were no significant differences between fast and slow wetting. Our results agree with previously published data that suggested an increase of aggregate stability after applying PAM to the soil (Shainberg et al.,. 1992; Aese et al., 1998; Levy and Miller, 1999; Mamedov et al., 2007), but disagree with others (Green et al., 2004). The results of the dispersion experiment for untreated and treated (0 and 50 kg ha-1 PAM, respectively) DF soil are presented in Table 8. The PAM treatment showed significantly lower dispersion values compared to control, indicating that PAM was effective in preventing clay dispersion.
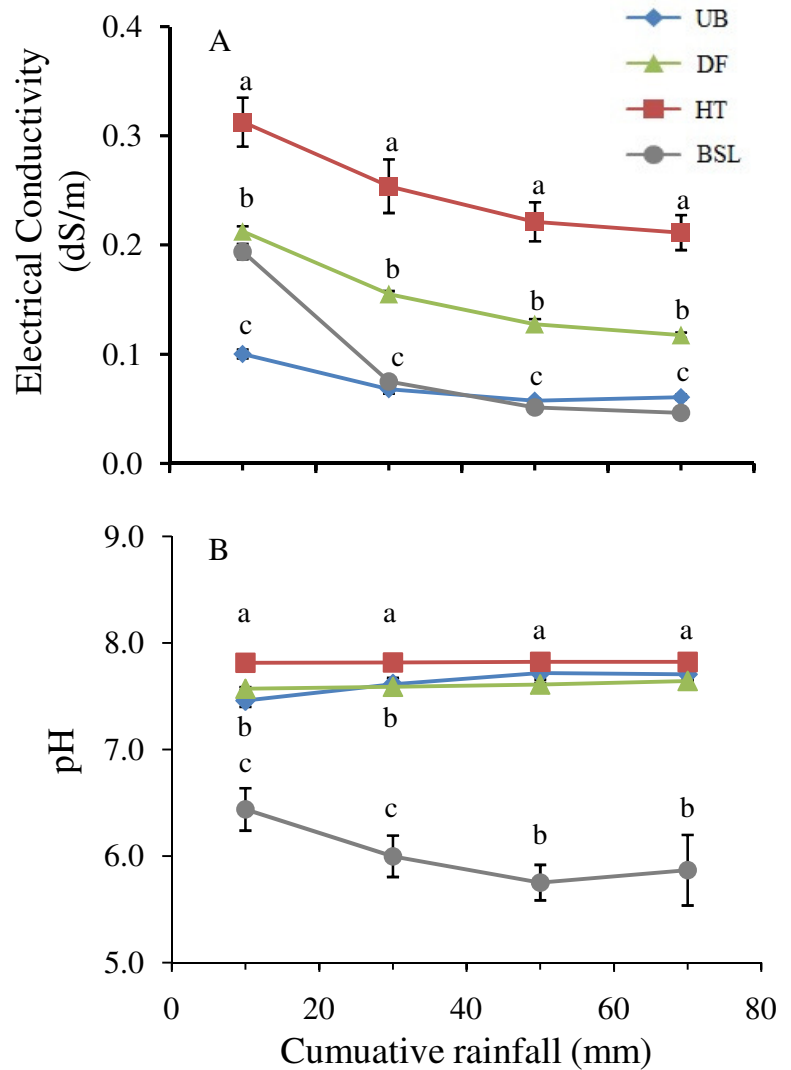

Figure 12: *EC (A) and pH (B) of the runoff collected from UB, DF, HT Pale rendzina fire treatments and BSL soil during the first storm in the rainfall simulator experiment; Different lowercase letters indicate significant differences between soils in each fractions. ($\alpha \leq 0.05$). Bars indicate one standard error.*

4.2.2 Discussion

In this chapter we tested the effect of applying granular PAM to the soil surface on IR, runoff and soil erosion on the Pale rendzina soils affected by different fire treatments, and on the burnt BSL soil (Tables 1 and 2), in laboratory experiments with disturbed soil and simulated rainfall. Few studies have tested the effect of granular PAM as a post-fire erosion control method and their conclusions were non-conclusive (Davidson[*] et al., 2009; Rough et al., 2007). Likewise, no published data was found for testing the PAM in laboratory controlled conditions using a

Figure 13: Mean weight diameter after fast and slow wetting and slaking value (SLV) of DF Pale rendzina soil. An asterisk indicates significant differences between fast and slow wetting in each treatment (α≤ 0.05).

Soil	SLV
Control	1.19
PAM 50 kg/ha	1.01
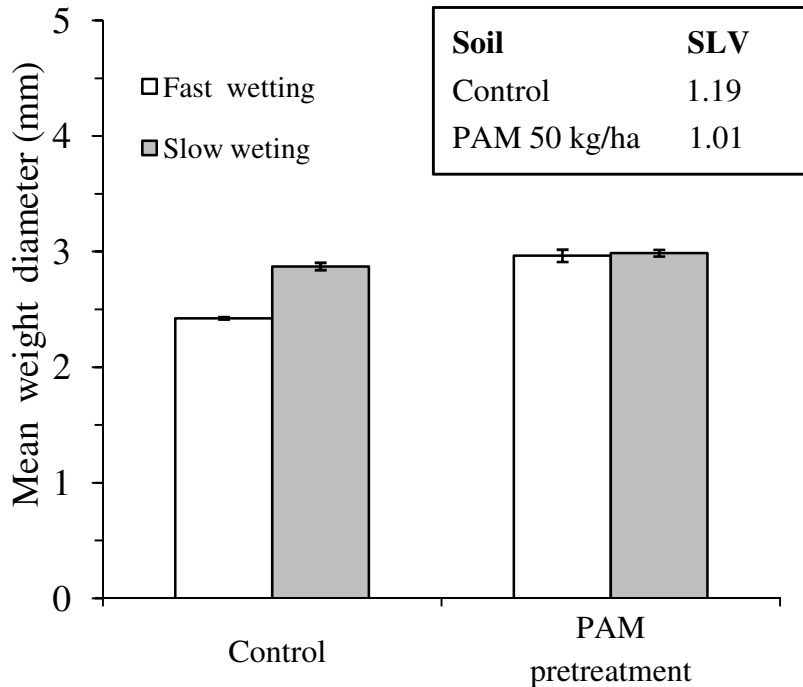
rainfall simulator on IR, runoff and soil loss of burnt soils, and after wetting and drying cycles. Therefore, the present chapter represents a unique contribution to the studies of soil protection after forest fires.

In general, runoff and IR values in all the PAM-treated soils tested in this experiment show similar behavior throughout the three rainstorms: in the 1st rainstorm, IR was decreased and runoff was increased compared to untreated control. However, in consecutive rainstorms, IR raised, causing the runoff amounts to drop to similar or sometimes significantly lower amounts compared to control (Figs. 7 - 10 and Table 6). As for soil loss, the UB, DF and BSL soils showed similar behavior throughout the three rainstorms: i.e., lower soil loss compared to untreated control in all rainstorms, although not always statistically significant (Table 6).

Table 8: *Dispersion value and electrical conductivity obtained during a dispersion test with the DF soil. Difference lowercase letters indicate significant differences between treatments.*

Treatments	Dispersion Value	Electrical conductivity
	---(%)---	---dS/m---
Control (0 kg PAM ha^{-1})	19.8 a	0.75 a
50 kg PAM ha^{-1}	3.0 b	0.75 a

Nevertheless, the HT PAM-treated soil differed from the rest by the higher soil loss amounts in the 1st rainstorm (Fig. 9, Table 6).

4.2.2.1 Effect of PAM on viscosity and quality of the runoff

In the 1st rainstorm, the contradictory results on the effect of PAM (increase in runoff but decrease of soil loss) in the UB, DF and BSL soils could be a result of the different mechanisms that occur simultaneously when PAM granules dissolve as the rain reaches the surface of the soil: an increase of soil solution viscosity, and an increase of aggregate stability and flocculation.

The dissolution of PAM and its interaction with soil solid particles enhances aggregate stability (Levy and Ben-Hur, 1998; Ben-Hur, 2006; Sojka et al., 2007). But at the same time, as PAM granules dissolve, there is an increase of the viscosity of soil solution and runoff (Fig. 11). During the beginning of the rain, PAM granules dissolved by rainwater increase the viscosity of the solution due to the long linear chains of PAM molecules (Malik and Letey, 1992; Ben-Hur and Keren, 1997). Hydraulic conductivity of porous media, and thus flow rate through it, is inversely proportional to fluid viscosity (Corey, 1977). Therefore, an increase in viscosity of the percolating solution can cause a decrease in the IR due to the reduction of the hydraulic conductivity of the soil (Malik and Letey, 1992; Ben-Hur and Keren, 1997; Ajwa and Trout, 2006). Ben-Hur and Keren (1997) suggested that high viscosity decreases the solution flow rate in the soil pores and consequently reduces IR, due to interactions of PAM molecules with soil particles within the soil profile. Furthermore, the authors postulate that these adsorbed molecules can cause stearic interference as a result of the adsorption of part of their segments to soil particles while other segments extend away from the surface as loops and tails, further limiting the penetration of other PAM molecules.

The differences in relative viscosity of the runoff of the PAM-treated soils (Fig. 11) could be attributed to different electrolyte concentrations in the runoff, since the viscosity of the PAM solution is affected by its electrolyte concentration (Ajwa and Trout, 2006). Ajwa and Trout (2006) found that treating a sandy loam soil with a PAM solution with high EC reduces the negative effect of PAM on FIR and hydraulic conductivity, due to decrease in viscosity of the leachate. It has been suggested that the presence of electrolytes cause the PAM molecules to get coiled up by the bridging effect of divalent cations, and coiled PAM molecules make the solution less viscous, and at the same time are less capable of clogging pores in the soil profile (Ajwa and Trout, 2006). The electrolyte concentration is represented as EC values of the runoff of the untreated control soils (Fig. 12). The latter data represents the potential effect of the electrolytes in the runoff solution on its viscosity with the addition of dissolved PAM molecules. Although having significantly different EC values (Fig. 12), the viscosity of the runoff of UB and DF soils was similar, probably due to the existence of a threshold EC value, below-which there is no difference in relative viscosity of the PAM solutions. The high EC in the runoff of the HT soil compared with the other soils could be the reason why the differences in relative viscosity of the runoff of PAM-treated and control soils are low (Fig. 11).

The higher viscosity of the BSL soil compared to the UB, DF and HT soils cannot be a result of differences in the electrolyte concentration, represented by EC values (Fig. 11D and 12A), since the EC of runoff of the BSL soil was similar to that of the UB and DF soils. We suggest that the higher relative viscosity of the PAM treated BSL soil compared to the UB, DF and HT soils was a result of different ion composition in the soil solution which affected the viscosity of the runoff. Although not statistically significant, the BSL soil had the lowest calcium and magnesium content from all the soils (data not shown). The reason for low calcium and magnesium is that the BSL soil originated by weathering processes of igneous granite rocks thus has low calcium and magnesium, compared to the Pale rendzina soil that calcareous sedimentary rocks are its parent material. Furthermore, the BSL soil is washed from its salts due to large amount of annual precipitation in Galicia, which also explains the low sodium content compared to the Pale rendzinas (data not shown) and its lower pH (Fig. 12B). Another difference that might exist in the ion composition of the BSL soil is the lower pH which indicates more protons (H^+ ions) in the soil solution (Fig. 12B). Lack of dissolved divalent cations and surplus of monovalent protons in the soil solution might enhance a more linear conformation to the PAM

molecules, preventing the decrease in the viscosity of the runoff solution. However, no reports were found about the effect of pH on the viscosity of PAM solutions and further research should be conducted on this matter. While runoff with dissolved PAM molecules flows over the soil, polymer molecules adsorb to soil particles on the surface and in pores in the soil profile. It can be assumed that more PAM molecules adsorb to the rendzina soil, rich with montmorillonitic clay particles compared to BSL soil which is rich with kaolinitic clay due to the significantly lower surface area of the latter soil, all the more so, considering its' initial lower clay content (Table 1). The result of all the aforementioned factors could have caused more PAM molecules to wash with the runoff, resulting in greater quantity of PAM molecules in the solution, increasing its relative viscosity to higher levels as compared to the rendzina (Fig. 11).

In all the PAM-treated soils, the decrease in viscosity during the rainstorm (Fig. 11) indicates that PAM molecules that are not adsorbed to soil particles, get transported downslope by the runoff. Furthermore, in all the PAM treated soils, the relative viscosity of the leachate was not significantly different than the one of the untreated controls (data not shown), since PAM molecules that are adsorbed to soil particles do not desorb to the percolating solution (Nadler et al., 1992). Furthermore, the lack of difference between the PAM and control treatments in the 2nd and 3rd rainstorms in the runoff (data not shown) is attributed to the adsorption of PAM molecules that are not washed away by the runoff in the 1st rainstorm, to soil particles during the first drying period. This interaction of PAM molecules with the soil particles, creating strong irreversible interactions that are preventing the PAM molecules' desorption and reaching to the soil solution again (Nadler et al., 1992).

4.2.2.2   Effect of PAM on IR and runoff amounts

The IR and runoff values obtained in PAM-treated Pale rendzina fire treatments in the three consecutive rainstorms could be explained as follows: In the 1st rainstorm, PAM granules are dissolved by the rain water. The dissolved PAM molecules interact with soil particles, increasing aggregate stability and flocculate the dispersed particles (Fig. 13 and Table 8, respectively). As a result, seal formation decreased in the PAM-treated soils. At the same time, the dissolution of PAM increases the viscosity of the percolating solution, compared to the control (Fig. 11). This higher viscosity decreases the IR of the soil and increases the runoff (Figs. 7A, 8A, 9A, and Table 6). Consequently, it can be assumed that the decrease in the IR in the 1st rainstorm (Figs. 7A, 8A and 9A) is a result of the increase in the viscosity of the runoff (Fig. 11)

rather than a formation of a seal on the soil surface (Yu et al., 2003). Ajwa and Trout (2006) conducted column experiment to measure hydraulic conductivity of a sandy loam soil applying different PAM solutions, and found that when PAM was applied with the leaching solution there was a negative effect on IR. These authors also found that flow rate of PAM solutions through uniform silica sand columns was strongly affected by PAM concentration. However, this negative effect disappeared when the columns were let to dry during 48 h before being leached with another 20 mm of DI water. Ajwa and Trout (2006) postulated that no remnant soluble PAM residues were present after the drying cycle. The reduction of viscosity in the last 40 mm (Fig. 11), however, does not cause an increase in IR in the Pale rendzina fire treatments (Fig. 7-9) as expected; resulting in low FIR values (Table 6). From these results it can be assumed that PAM molecules that are not washed by the runoff get adsorbed to soil particles on the soil surface and in the soil profile, cause an increase in the steric interference within pores (Malik and Letey, 1991), and keep water from infiltrating.

As opposed to the rendzina fire treatments, IR of the PAM–treated BSL soil increases with the increase in rainfall depth throughout the 1st rainstorm (Fig. 10A). We postulate that this increase in IR was caused by the some or all of the following factors: (i) as described in chapter 4.2.2.1, the higher viscosity of the BSL soil which was caused by higher amounts of PAM in the runoff (as compared to the runoff of PAM-treated rendzina soil), caused most of the PAM molecules that were not adsorbed to soil particles to be washed away in the first 40 mm of rainfall; (ii) BSL soil is coarse textured which can lead to high IR. Thus it is expected that percolating PAM solution would pass through pores more freely, as compared to the rendzina soils, all the more so with the decrease in the relative viscosity of the runoff (Fig. 11). The fact that the viscosity of the leachate in the PAM-treated BSL soil was not different to that of DW (data not shown), indicates that most of the PAM molecules in the percolating water were adsorbed to the soil particles and were not leached further down through the soil profile; (iii) in low pH (under pH 7), anionic PAM molecules are adsorbed to kaolinitic clay particles through hydrostatic interactions with the positive broken edges of the clay (Peng and Di, 1994) , hence it is likely that PAM molecules on the soil surface and within the soil profile were fully adsorbed to the clay, leaving no edges that may cause steric interference to percolating water (Malik and Letey, 1991). The decrease in the relative viscosity of the runoff caused the rise in IR to levels close to untreated control towards the end of the 1st rainstorm (Fig. 10).

The drying period between the 1st and 2nd rainstorms caused PAM molecules that were not washed away by the runoff to adsorb to soil particles, and therefore relative viscosity of runoff and leachate showed no differences in the PAM-treated and control soils in the 2nd and 3rd rainstorms in all soils (data not shown). These interactions result in a further increase of aggregate stability in the PAM-treated soil (Fig. 13, Table 8), and therefore aggregate breakdown and seal formation are reduced (Shainberg et al.,. 1992; Aese et al., 1998; Levy and Ben-Hur, 1998; Levy and Miller, 1999; Ben-Hur, 2006; Mamedov et al., 2007; Sojka et al., 2007). As a result of the irreversible interaction of PAM molecules and soil particles (Nadler et al., 1992), PAM solubility decreases, and so does the viscosity of the percolating solution during the following rainstorms (data not shown). In the 2nd and 3rd rainstorms, the increase of aggregate stability (Fig. 13, Table 8) prevents the formation of a seal in the soil surface, and this phenomenon, in addition to the decrease of the viscosity of the percolating solution, reverses the effect of PAM on IR and runoff (Fig. 7-10 and Table 6; Ajwa and Trout, 2006). In all soils, total runoff of the PAM treatment significantly decreased in the 2nd rainstorm to reach similar or lower values as the control (Table 6). Finally, after another drying treatment, in the 3rd rainstorm the PAM treated soils showed higher IR and higher FIR values and lower runoff values than the control (Figs. 7A, 8A, 9A, 10A, and Table 6), probably as a consequence of persistence of the stabilization mechanism .

It must be noted that the IR during the 2nd rainstorm in the DF soil reacted differently as compared to the other soils. During the 2nd rainstorm, IR of the PAM-treated DF soil increased to reach values higher than those of the control at the end of it (Fig. 8A) as a result of higher aggregate stability and a decrease of the solution viscosity. The low initial values in this rainstorm might be a result of remnant PAM molecules, which affect the hydraulic conductivity in the PAM-treated soil and were not prominent enough to be measured in the viscometer.

4.2.2.3   Effect of PAM on soil loss

As noted before, during the 1st rainstorm the dissolution of PAM granules increases the viscosity of the runoff, decreases IR and increases runoff amounts in all the soils (Figs. 7A, 8A, 9A, 10A, 11 and Table 6). The high amounts of runoff would have been expected to cause a significant amount of soil erosion in the PAM-treated soils. However, there is a decrease in soil loss in the PAM-treated UB, DF and BSL soils (Figs. 7B, 8B, 10B and Table 6). This is attributed to the increased viscosity of the runoff (Fig. 11) and the PAM-induced flocculation

and increase in soil aggregate stability. The viscous runoff reduce soil loss in the aforementioned soils due to several reasons: (i) long chains of the PAM molecules interact with distant soil particles increasing their resistance to splash by raindrop impact and detachment by runoff (Yu et al., 2003) (ii) the high viscosity of the runoff causes it to flow more slowly on the soil surface, reducing its erosive effects and transport capacity; and (iii) there is an increase in the flocculation of soil particles in the runoff mediated by PAM molecules (Shainberg et al., 1990; Shainberg and Levy, 1994; Ben-Hur and Keren, 1997).

While in the PAM treated UB and DF soils loss amounts were low and steady or with slight increase, respectively, the BSL soil showed a different pattern in the 1st rainstorm (Fig. 10). As in the former soils, soil loss in the PAM treated BSL soil was low in the first 40 mm despite the high runoff amounts (Table 6), which is attributed to the aforementioned viscosity mechanism. In the second half of the 1st rainstorm, the increase in the IR (Fig. 10A; and as detailed in Chapter 4.2.2.2), resulted in lower runoff amounts and a decline in soil loss (Fig. 10B) reaching basal values, and resulting in low significant total soil loss (Table 6).

In the 1st rainstorm, the PAM treated HT soil showed also a decrease in IR, followed by increase in the runoff (Fig. 9A, Table 6). However, in the HT soil, the pattern of soil loss response to PAM application was different: while in the PAM treated UB, DF and BSL soils there were lower total soil loss values during the 1st rainstorm compared to their respective untreated soils (Table 6), in the HT soil this reduction was not observed, and conversely, there was an increase in soil loss with PAM application (Table 6). The effectiveness of dry PAM to decrease soil erosion may depend on the balance between the positive effect of PAM on flocculation and aggregation and the adverse impact of the viscosity of the soil solution on soil permeability and runoff generation. The untreated HT soil showed high IR and low runoff in the 1st rainstorm compared to the other two untreated rendzina soils (Table 4) due to its increase in structural stability with the thermal treatment (as discussed in chapter 3.2). As a consequence, soil loss was low (Table 4). As in the other soils, the addition of PAM to the HT soil increased the viscosity of the runoff (Fig. 11) and reduced the IR (Fig. 9A) compared to the untreated soil. As a result, there was higher runoff yield in the PAM-treated soil than in the control, as described in chapter 4.2.2.2. Nevertheless, runoff from the PAM-treated HT soil had lower relative viscosity than the runoff from UB, DF and BSL soils, and this is attributed to the higher electrolyte concentration (Fig.11). The aforementioned lower relative viscosity of the runoff

could have resulted in higher flowing speed, increased erosive forces and higher sediment transport capacity. Increased runoff but with low viscosity resulted in higher soil loss in the PAM-treated than in the control soil (Table 6). In other words, it can be summarized that the relative viscosity of the runoff from PAM treated HT soil is high enough to cause a reduction in IR and an increase in runoff compared to control, but it is not high enough to cause a reduction in the erosivity and transport capacity of the runoff.

The results from the slaking test (Fig. 13) in this work prove that aggregates that were exposed to high concentrated viscous PAM solution at the soil surface remain strong after a drying cycle due to adsorption of PAM molecules on their exterior surface, thus resisting the slaking forces of fast wetting (Mamedov et al,. 2007). Furthermore, the external coating of the aggregates that PAM produces (Mamedov et al,. 2007) allows them to withstand dispersion processes that can occur when the soil is exposed to rainwater with low electrolyte concentration, as proved by the dispersion test (Table 8). The stronger PAM-induced aggregate stability reduces the amount of detached small particles that can be transported by the runoff. Moreover, aggregates that break down in the 1st rainstorm by raindrop impact get flocculated quickly and form strong aggregates due to the binding capability of PAM molecules (Ben-Hur and Keren, 1997). These results agree with those of Ben-Hur and Keren (1997), who found that three types of polymer formulations had a positive effect on flocculation-aggregation processes compared to control. Thus, after the drying cycle, both newly formed and original aggregates are strong and resist the slaking forces caused by fast wetting, limiting seal formation, causing an increase in IR, a reduction in runoff and in soil loss, in all soils (Table 6). Moreover, even if not always statistically significant, this stabilization mechanism continues to affect the FIR, runoff and soil loss also in the 3rd rainstorm (Fig. 7-10; Table 6).

4.3   Part III: Field experiment

4.3.1   Results

4.3.1.1   Effects of different PAM rates on runoff and soil loss in field conditions

In the present work, a field experiment with runoff plots was conducted to test the effect of dry PAM on runoff and soil loss of a rendzina affected by forest fire and subjected to natural rainfall. Cumulative runoff (in depth units) and cumulative soil loss during the winter 2009-2010 for the three treatments as functions of cumulative rainfall are presented in Figs. 14A and 14B, respectively. Significant linear regressions were observed in all three treatments for the runoff and soil loss (Figs. 14A and 14B).

The total runoff and soil loss measured during the winter 2009-2010 for each treatment are presented in Table 9. Total runoff of the whole season did not differ much between treatments and was not found significantly different (Table 9). However, a clear trend is visible throughout most of the precipitation period, with the 50 kg ha^{-1} PAM treatment showing the lowest runoff yield (Fig. 14A, Table 9). A clear trend of reduction of soil loss with increasing PAM dose application was observed throughout the winter: an application of 25 kg ha^{-1} reduced soil loss by 23% with respect to the control plots, whereas an application of 50 kg ha^{-1} PAM showed a 50% reduction in total soil loss (Fig.14B and Table 9). However, the differences found were not statistically significant (Table 9). The lack of significance between the treatments is attributed to the spatial heterogeneity of the forest floor (Kutiel et al., 1995; Lavee et al., 1995) and of the effects of the wildfire, and the insufficient number of replicates for each treatment.

Relative viscosity measurements of the runoff were conducted throughout the winter and were found to be close to the viscosity of water (resulting in relative viscosity values close to 1), except in the 1st rainstorm, where relative viscosity values were 1.18, 1.09 and 0.97 for 50 kg ha^{-1}, 25 kg ha^{-1} and control, respectively.

4.3.1.2   Effects of PAM on aboveground biomass and seed production

Post-fire vegetation recovery is an important process which is a major concern to forest management. Several post-fire mitigation strategies also include seeding of local plants species (as reviewed by Robichaud et al., 2010). The effect of the two PAM treatments (25 and 50 kg ha^{-1}) on aboveground biomass and seed production compared to control is presented in Fig. 15A and 15B, respectively. No significant differences were found between the effect of the PAM

treatments and the control neither in aboveground biomass production, nor in aboveground seed production.
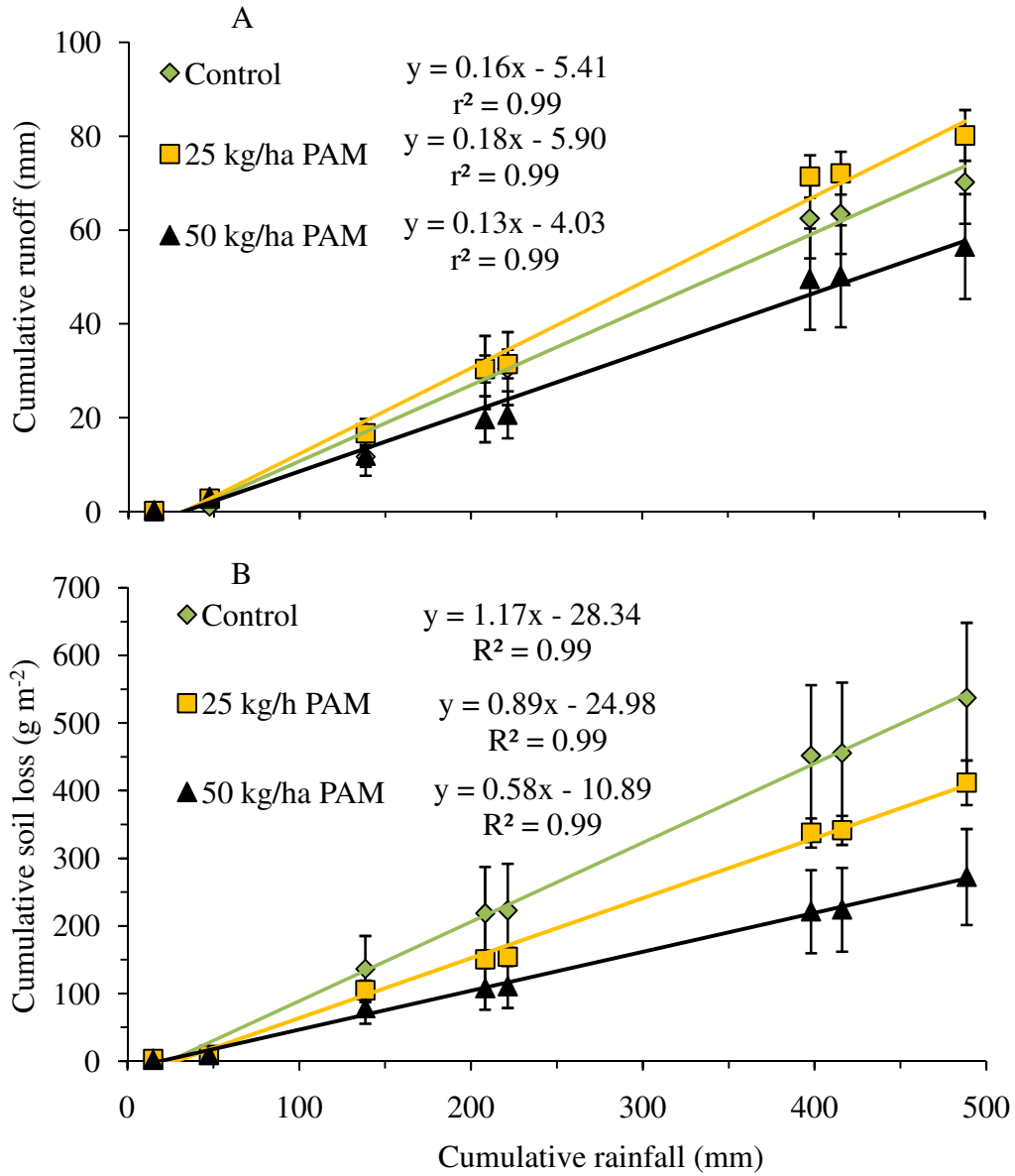

Figure 14: (A) Cumulative runoff and (B) cumulative soil loss collected during the rainy season (winter 2009-2010) in the field experiment in Birya forest, as function of cumulative rainfall. Bars indicate one standard error.

Table 9*: Total runoff and soil loss collected during the winter 2009-2010 in control, 25- and 50 kg ha^{-1} PAM treatments of the field experiment in Biryia forest. Numbers in parenthesis represent the runoff coefficient (i.e. runoff as percentage of total rainfall).*

	Control	25 kg ha^{-1}	50 kg ha^{-1}
	--------------------------------- mm ---------------------------------		
Runoff yield	70.2 (14%)	80.2 (16%)	56.5 (12%)
	--------------------------------- g m^{-2} ---------------------------------		
Soil loss yield	536.8	411.9	272.5

4.3.2   Discussion

4.3.2.1   Effects of different PAM rates on runoff and soil loss in field conditions

Few studies have been made on testing the effect of dry PAM on soil erosion after forest fires in field conditions. These studies have tested the application of PAM alone (Rough, 2007; Davidson et al., 2009) or combined with straw mulch (Davidson et al., 2009). Davidson et al., (2009) conducted a three year experiment in which they compared the effects of the application of PAM and straw mulch on soil erosion. The experiment took place after the 2004 Red Bull Fire in Utah, after the area has been seeded with local vegetation. The four treatments that were tested were: (i) Control burnt soil; (ii) PAM; (iii) PAM+straw mulch; and (iv) straw mulch on slopes of 16%, 33%, 25% and 20%, respectively. The PAM was spread as recycled paper pellets containing various MW PAM with the average MW of 18×10^6 Da, at a rate of 8 kg ha^{-1}. The results showed less soil erosion from the PAM and PAM+mulch treatments compared with mulch and control treatments, but with no significance in the mean values. The researchers explained that although the results showed no significance, the PAM and PAM+mulch treatments were on the steepest slopes, hence their soil erosion rates were expected to be the highest. On the other hand, Rough (2007) conducted a three year experiment using granular and dissolved PAM, and did not find a constant trend in reduction of soil erosion.
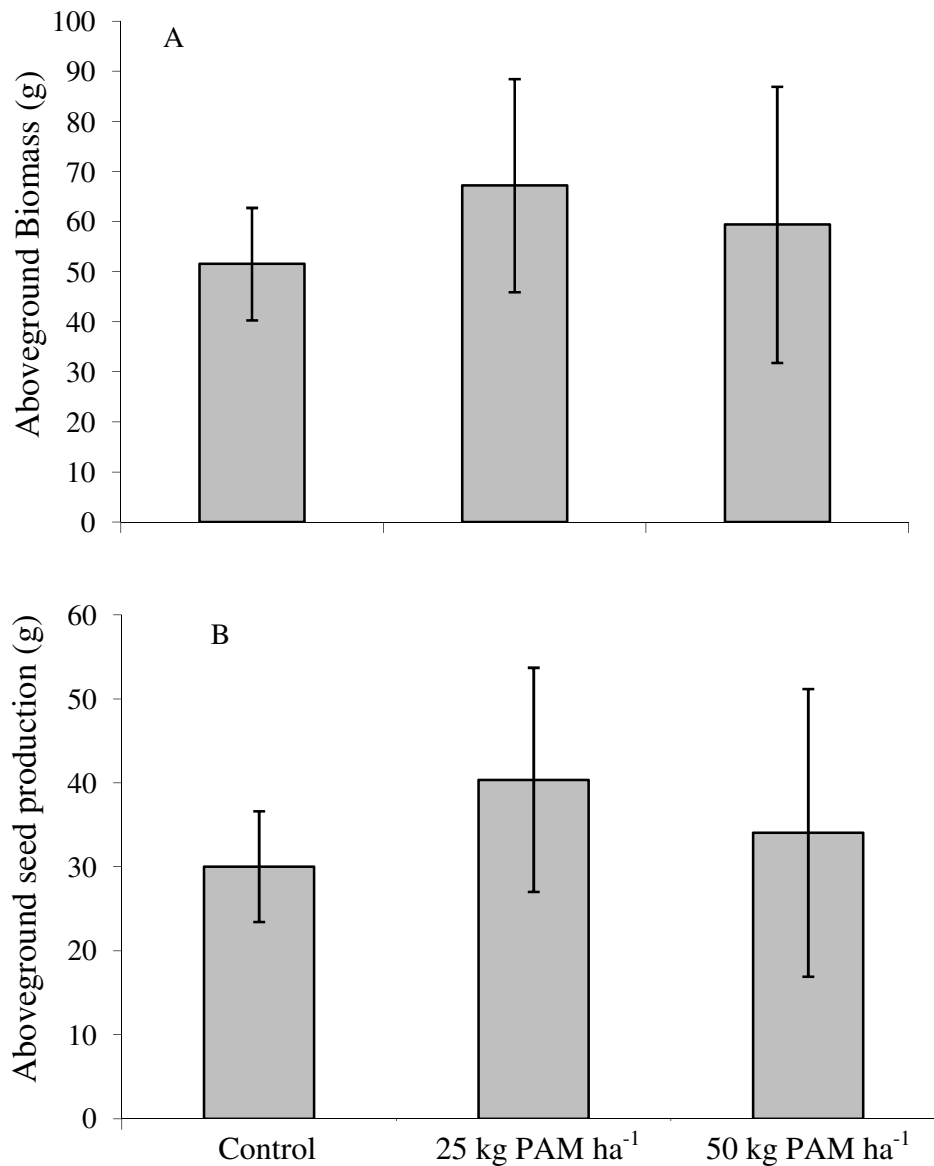
Figure 15: *(A) Aboveground production of biomass and (B) seed production of wheat (Triticum aestivum, L.) from plots treated with 0, 25 and 50 kg PAM ha⁻¹. Bars indicate one standard error.*

Total annual runoff and soil loss yield from the untreated control (Table 9) were 70.2 L m^{-2} and 536.8 gr m^{-2}, respectively, with seasonal rainfall of 488mm measured since the beginning of the experiment (Fig. 14). Inbar et al. (1998) conducted a long term experiment in Mt. Carmel, Israel, after a severe wildfire that occurred there in 1989. After the fire, 15 plots of 200m^2-400m^2 were built in order to record runoff and sediment from the burnt site. The plots were placed in north and south facing slopes, with 20%-30% slope and on top of Terra rosa soil.

It was found that runoff and soil loss amounts in the first year after the fire were 4.1L m^{-2} and 90.3gr m^{-2} for the north facing slope, respectively, and 5.4L m^{-2} and 374.5gr m^{-2} for the south facing slope, respectively, with an annual rainfall of 714mm (Inbar et al., 1997, 1998; Wittenberg and Inbar, 2009). In the current experiment, runoff was considerably higher per m^2 compared to the Inbar et al. experiment. However, on average, for every liter of runoff, the low-moderately burnt Pale rendzina soil yielded 7 gr of sediment compared to 22gr and 69gr from the north and south facing slopes, respectively, which was recorded by Inbar et al. (1998). The differences in the results between the current experiment and the results presented by Inbar el al. (1998) could have been a result of the following reasons: (i) different soil type and properties; (ii) different slopes and aspects (i.e. north and south vs. west); (iii) difference in surface cover (i.e. presence of ash and burnt trees in the Inbar et al. experiment) ; (iv) different fire severity; and (vi) different rainfall amounts and intensities; and (vii) different plot scale (Shakesby and Doerr, 2006).

In an experiment conducted in burnt forest plantation in Yoqneam forest, Kutiel and Inbar (1993) measured runoff and sediment yield from a burnt and an adjacent unburnt plot, with an average area of 155.5m^2 on an average slope gradient of 43% with a north-western aspect. The fire took place in October 1988 and was of light-moderate severity. The soil in the experimental site was Pale rendzina (Lithic Xerothenth), with 56% CaCO$_3$ and 6.8% OM. The measurements were taken for two months, during which 8 rainstorms with total of 331mm precipitated. Runoff and sediment yields measurements from the burnt site were 0.168L m^{-2} and 0.046gr m^{-2}, respectively, a very low value compared to the current research (Kutiel and Inbar, 1993). One of the reasons that could have caused the large differences was differences in surface cover. While in the Yoqneam forest fire there was high rainfall interception potential due to the fact that only the understory was burnt and the canopy remained undamaged (Kutiel and Inbar, 1993), in Birya forest the trees and deposited ash were removed from the area of the plots, leaving the soil bare. Another reason for the aforementioned differences could have been the different scale.

The application of PAM did not have a significant effect on runoff, although the 50 kg ha^{-1} application rate resulted in the lowest runoff yield in the end of the season (Table 9). Moreover, a clear trend is visible in which the 50 kg ha^{-1} treatment showed the lowest runoff throughout the rainy season (Fig. 14A). A clear trend is also apparent in soil loss yield, whereas 50 kg ha^{-1} was the lowest throughout winter (Fig. 14B). In this case, although not significantly,

differences were more conspicuous, whereas 50 kg ha^{-1} PAM application reduced soil loss in 50% whereas 25 kg ha^{-1} PAM application reduced soil loss by 23%, in the end of the rainy season (Table 9).

From the results as presented in Figure 14, it can be inferred that the increase in viscosity in the 1st rainstorm (i.e. 1.18, 1.09 and 0.97 for 50 kg ha^{-1}, 25 kg ha^{-1} and control, respectively) did not have any effect on runoff yield in that rainstorm (Fig. 14A) as compared to results from the rainfall simulator experiments (Figures 7, 8, 9 and Table 6), and neither in a whole season perspective, whereas similar amounts of runoff were recorded at the end of the winter to all the treatments (Table 9).

The 1st rainstorm event recorded in the field experiment (on the 25th of November, 2009) had low rainfall depth (16 mm) precipitated over 24 hours, which resulted in very low runoff in all treatments (Fig. 14A). We postulate that this low precipitation event was long lasting enough to cause the complete dissolution of the PAM granules, but was not intense enough to produce the large, viscous runoff amounts that was recorded in the same rainstorm in the rainfall simulator experiment (Figures 7, 8, 9 and Table 6). The drying period after the 1st rainstorm (11days) allowed the PAM molecules to adsorb to soil particles and stabilize the soil in such a way that lasted the rest of the winter (Figure 14A), indicating that the stabilization mechanism that was identified in the 2nd and 3rd rainstorms in the rainfall simulator experiments was dominant for the rest of the season.

4.3.2.2 Effects of PAM on aboveground biomass and seed production

The recovery of the vegetation in the first year after the fire is critical for establishment of initial vegetation cover that provide a source for next year seeds and plant litter cover. In the previous chapter we discussed the effect of PAM as a post-fire soil erosion mitigation treatment. While several studies have dealt with the effect of PAM on seedling emergence (Cook and Nelson, 1986; Rubio et al., 1989; Steinberger and West, 1991, Rapp et al., 2000) and/or biomass production (Stern et al. 1992, Rapp et al., 2000), few have checked the effect of PAM on the vegetation recovery after forest fire (Davidson et al., 2009). PAM can effect post-fire plant regeneration: (i) by directly affecting the vitality of the seeds and seedlings; and (ii) by indirectly effecting seed transport by runoff, water availability in rhizosphere and crust formation.

Davidson et al. (2009) studied the effect 2 kg ha^{-1} granular PAM and straw mulch combinations on soil erosion and the recovery of the vegetation in a burnt forest area seeded with

local vegetation. The authors found a positive effect of the PAM treatment (with or without straw mulch) on vegetation recovery and surface cover in the first year after the fire. These results indicate that besides reduction in soil erosion, PAM promotes faster vegetation recovery in the critical first year after the fire.

In this study, no significant differences in aboveground biomass and seed production were found between the three treatments (Fig. 15). From these results it can be concluded that the PAM application does not influence the checked criteria, regardless of its application rate.

Further research is needed to investigate the effects of PAM treatments on vegetation recovery in burnt forests in Israel considering its low annual rainfall amount, and its long annual drought period. Furthermore, the ecological effect of PAM on species diversity and richness of flora, fauna and microorganism and their effect on secondary succession processes should be investigated in future studies.

5   Conclusions

Different changes in soil properties may exist in a burnt location due to spatial variability in fuel load, soil properties and depend highly on climatic conditions and topography. The different variability in fire conditions may influence the underlying soil differently, resulting in a mosaic pattern of different effects (Kutiel et al., 1995; Lavee et al., 1995). In the present study, we tested the effect of different fire treatments on the soil properties of a Pale rendzina soil from the Galilee. It was found that different fire treatments make changes in the physical and chemical properties of the Pale rendzina soil. As a result of a prolonged exposure to moderate temperatures treatment (i.e. HT soil), these changes cause the structural stability to increase and resist the destruction of aggregates by raindrop impact, by slaking during fast wetting, and by dispersion of clay particles when wetted by rainwater. The stabilization was found to be a result of changes in physical and chemical soil properties. Furthermore, the mineralization of organic compounds from the combustion of the vegetation and litter layer chemical changes resulted in concentrations of electrolytes in the soil solution higher than the flocculation values of the clay. Therefore, clay dispersion was reduced. Hence, when exposed to raindrop impacts, the HT treatment showed high IR, and very low runoff and soil loss values compared to UB soil. In case of moderately affected soil, no structural changes were found. Furthermore, aggregate stability tests (i.e. slaking and dispersion) did not show changes in aggregate stability of DF soil

compared to the UB soil. However, although lower than HT soil, DF soil showed higher IR than UB soil, which resulted in lower runoff and soil loss. Further research should be conducted to explain the causes of the differences between DF and UB soils.

The stabilization effects of fire on Pale rendzina decreased in consecutive rainstorms, and as a result lower IR and higher runoff and soil erosion where found as the number of rainstorms that hit the soil surface increased. However, even though there was an increase of aggregate stability with fire, a post-fire net increase in runoff and soil erosion from one rainstorm to the next was observed, possibly due to lack of surface cover. Therefore, the implementation of post-fire erosion control measures after a wildfire is crucial. We tested the usage of PAM, which have been proven useful in reducing soil loss in arable lands and stabilizing steep slopes, as an alternative for post-fire erosion control method.

In laboratory rainfall simulator experiments, the application of 50 kg ha^{-1} PAM to UB and DF soils increased runoff during the 1st storm, but after a drying period runoff decreased compared to a control treatment. Regarding soil loss, the same PAM application reduced soil erosion in the UB and DF soils . Two different mechanisms were identified as responsible for this response of runoff and soil loss:

(i)   Increase of soil solution viscosity. – PAM dissolved by rainwater during the 1st rainstorm increases the viscosity of the runoff, reducing its velocity while flowing on top of the soil and consequently its erosivity and transport capacity. The result is a decrease in IR and an increase of runoff, but significantly lower soil loss. After a drying period, PAM gets irreversible attached to soil particles, and therefore runoff viscosity returns to values similar to those of control treatments.

(ii)   Increase of aggregate stability. –PAM molecules that were not washed away by the runoff get adsorbed to soil particles, promote clay flocculation and stabilize the soil surface against sealing, and increasing IR and reducing runoff and soil erosion. The interactions soil particle-PAM molecules are enhanced during the drying period between the 1st and the 2nd rainstorm.

In HT soil, the high EC of the runoff resulted in lower solution viscosity when PAM dissolved. Therefore, no reduction in soil loss was observed during the 1st rainstorm. However, in consecutive rainstorms, IR increased, and runoff and soil loss decreased as a consequence of the improved aggregate stability by PAM molecules.

The reduction of soil loss as a result of PAM application to a Pale rendzina were observed too in a sandy loam soil from Spain that was treated by a similar prolonged exposure to moderate temperatures treatment as the HT soil (BSL), which has different properties as mineralogy, texture, and organic matter content. Therefore, PAM can be considered as an effective erosion control measure in a wide range of soil conditions.

PAM application was also effective in reducing soil loss in natural conditions. In a field experiment conducted on the Pale rendzina after a forest fire, soil erosion was reduced during the winter 2009-2010 as PAM application rate increased. It was found that 50 kg ha^{-1} granular PAM application was effective in reducing runoff and soil loss, reducing the latter by 50% compared to untreated control. An increase of viscosity during the 1st storm was observed, which disappeared after the 1st drying period.

A possible limiting factor to establishment of vegetation is availability of water in the soil profile. In the same field experiment, no effect of PAM was observed on annual grass growth during the rainy season. In the rainfall simulator experiment, the 50 kg ha^{-1} PAM treatment had shown lower IR values in the 1st rainstorm due to increased viscosity of rainwater that dissolved the PAM granules.. However, in the field experiment, better IR values were measured during the whole winter in the 50 kg ha^{-1} PAM treatment as a result of a better soil structure and a less developed seal compared to the control.

Several studies have shown that PAM increase vegetation productivity due to its effectiveness in increasing the resistance to seal formation which can be an impermeable barrier to seedling emergence (Cook and Nelson, 1986; Rubio et al., 1989), while other studies did not see improvement in seedling emergence compared to control even though the latter had a well-developed seal on the surface (Steinberger and West, 1991). In the field experiment the total runoff yields that were measured during the whole rainy season were similar between all treatments, whereas the 50 kg ha^{-1} treatment had the lowest values. From these results it can be assumed that in all treatments, similar amounts of water reached the rhizosphere area for plant usage and can explain the absence of differences in aboveground biomass and seedling production between PAM treatments and control. The differences in runoff and soil erosion indicate that a less developed seal was formed in the 50 kg ha^{-1} treatment, but no significant differences in the biomass and fruit yield were found between the treatments, which indicates

that the seal formed at the soil surface did not affect the seedling emergence regardless of the treatment, which may be a result of the relatively high stability of the forest Pale redzina soil.

From the results of this preliminary study, we suggest the following:

(i) Post fire erosion control methods should be implemented before the beginning of the rainy season despite the stabilization effects of the fire.

(ii) The usage of PAM should be implemented in area that no vegetation cover was left.

(iii) PAM can be used as a post-fire rehabilitation treatment however, further research should be conducted

Further research that should be tested in the following topics:

(i) Testing the interactions of ash-PAM and their effect on IR-runoff-soil loss relationships.

(ii) Testing granular PAM application in larger scales, i.e. slope and catchment scale.

(iii) Testing PAM application on different soils, such as Terra rosa, which dominants large parts of the forest in Israel.

(iv) Investigate vegetation recovery and secondary succession processes as affected by PAM application

(v) Investigate the effect of PAM on micro and macro fauna in the post-fire treated area.

6   Acknowledgments

7   References

טסלר, נ., גרינבאום, נ., ויטנברג, ל. 2008. תהליכי השתקמות ארוכי טווח בתכונות הכימיות של הקרקע לאחר שרפות יער בכרמל, 1983-2006. אופקים בגאוגרפיה. 71:137-118.

Aase, J.K., D.L Ljornebrg, and R.E Sojka. 1998. Sprinkler irrigation runoff and erosion control with polyacrylamide - laboratory studies. Soil Sci. Soc. Am. J. 62:1681-1687.

Agassi, M., I. Shainberg, and J. Morin. 1981. Effects of electrolyte concentration and soil sodicity on the infiltration rate and crust formation. Soil Sci. Soc. Am. J. 45:848–851.

Ajwa, H.A., and T.J. Trout. 2006. Polyacrylamide and water quality effects on infiltration in sandy loam soils. Soil Sci. Soc. Am. J. 70:643–650.

Allison, L.E. 1965. Organic carbon. In 'Methods of soil analysis. Chemical and microbiological properties'. Agronomy No. 9, Part 2. (Eds CA Black, DD Evans, JL White, LE Ensminger, FE Clark) pp. 1367–1378. (American Society of Agronomy, Inc.: Madison, WI).

Allison, L.E., and C.D. Moodie. 1965. Carbonate. In 'Methods of soil analysis. Chemical and microbiological properties'. Agronomy No. 9, Part 2. (Eds CA Black, DD Evans, JL White, LE Ensminger, FE Clark) pp. 1379–1396. (American Society of Agronomy, Inc.: Madison, WI).

Aly, S.M., and J. Letey. 1988. Polymer and water quality effects on flocculation of montmorillonite. Soil Sci. Soc. Am. J. 52:1453–1458.

Andreassian, V. 2004. Water and forest: from historical controversy to scientific debate. J. Hydrol. 29:1–27.

Assouline, S. 2004. Rainfall-induced soil surface sealing: A critical review of observations, conceptual models, and solutions. Vadose Zone J. 3:570–591.

Assouline, S., and Y. Mualem. 2000. Modeling the dynamics of soil seal formation: Analysis of the effect of soil and rainfall properties. Water Resour. Res. 36:2341–2349.

Badía, D., and C. Martí. 2003. Plant Ash and Heat Intensity Effects on Chemical and Physical Properties of Two Contrasting Soils. Arid Land Research & Management. 17(1): 23-41.

Baver, L.D., W.H. Gardner , and W.R Gardner .1972. Soil Physics. John Wiley, New York.

Ben-Hur, M., I. Shainberg, D. Bakker, and R. Keren. 1985. Effects of soil texture and $CaCO_3$ content on water infiltration in crusted soils as related to water salinity. Irrigation Science. 6:281–284.

Ben-Hur, M., I. Shainberg, and J.Morin. 1987. Variability of infiltration in a field with surface-sealed Soil. Soil Science Society of America Journal. 51(5): 1299-1302.

Ben-Hur, M., M. Malik, J. Letey, and U. Mingelgrin. 1992. Adsorption of polymers on clays as affected by clay charge and structure, polymer properties, and water quality. Soil Science. 153: 349–356.

Ben-Hur, M., and R. Keren. 1997. Polymer effects on water infiltration and soil aggregation. Soil Sci. Soc. Am. J. 61:565–570.

Ben-Hur, M. 2006. Using synthetic polymers as soil conditioners to control runoff and soil loss in arid and semi-arid regions—a review. Australian Journal of Soil Research. 44:191–204.

Ben-Hur, M. 2008. Seal formation effects on soil infiltration and runoff in arid and semiarid regions under rainfall and sprinkler irrigation conditions. In: Zereini, F., Jaeschke, W. (Eds.), Climatic Changes and Water Resources in the Middle East and in North Africa. Springer-Verlag, New York, pp. 429–452.

Ben-Hur, M., G. Yolcu, H. Uysal, M. Lado, and A. Paz. 2009. Soil structure changes: aggregate size and soil texture effects on hydraulic conductivity under different saline and sodic conditions. Australian Journal of Soil Research. 47: 688–696.

Ben-Hur M, C. Fernandez, S. Sarkkol, and J.C. Santamarta Cerezal. 2011. Overland flow, soil erosion and stream water quality in forest under different perturbations and climate conditions. In: Bredemeier M, S, Cohen, D.L. Godbold, E. Lode, V. Pichler, and P. Schleppi (eds) Forest management and the water cycle: an ecosystem-based approach. Series: Ecological Studies, vol 212. Springer, Berlin, pp 263–289. doi:10.1007/978-90-481-9834-4_15.

Benavides-Solorio, J., and L.H. MacDonald. 2005. Measurement and prediction of post-fire erosion at the hillslope scale, Colorado Front Range. International Journal Of Wildland Fire. 14:457–474.

Bresson, L.M., and J. Boiffin. 1990. Morphological characterization of soil crust development stages on an experimental field. Geoderma. 47:301–325.

Brock, J.H., and L.F. DeBano. 1982. Runoff and sedimentation potentials influenced by litter and slope on a chaparral community in central Arizona. p. 372–377. In C.E.C. Conrad and W.C.

Oechel (ed.) Proc. Symp. Dynamics and Manage. of Mediterranean-type Ecosystems. Gen. Tech. Rep. PSW-58. U.S. For. Serv., Pac. Southw. For. Range Exp. Stn., Berkeley, CA

Bruijinzeel, L.A. 2004. Hydrological functions of tropical forests: not seeing the soil for the tree?. Agri Ecosyst Environ. 104:185–228.

Calder, I.R. 2007. Forests and water – ensuring forest benefits outweigh water costs. For Ecol Manage. 251:110–120.

Campbell, G.S., J.D. Jungbauer, Jr., W.R. Bidlake, and R.D. Hungerford.1994. Predicting the effect of temperature on soil thermal conductivity. Soil Science. 158:307–313.

Cerda, A., and S.H. Doerr. 2008. The effect of ash and needle cover on surface runoff and erosion in the immediate post-fire period. Catena. 74:256-263.

Cerda, A., and P. Robichaud. (eds). 2008. Fire Effects on Soils and Restoration Strategies .Science Publisher: Enfield, New Hampshire, USA.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. Oecologia. 143: 1–10.

Chaney, K., and R.S. Swift. 1984. The influence of Organic Matter on Aggregate Stability in some British Soils. Journal of Soil Science. 35 :223-230.

Chapman, H.D. 1965. Cation-exchange capacity. In 'Methods of soil analysis. Chemical and microbiological properties'. Agronomy No. 9, Part 2. (Eds CA Black, DD Evans, JL White, LE Ensminger, FE Clark) pp. 891–901. (American Society of Agronomy, Inc.: Madison, WI)

Chen, Y., J. Tarchitzky, J. Brower, J. Morin, and A. Banin. 1980. Scanning electron microscope observations on soil crusts and their formation. Soil Sci .130:49–55.

Collis-George, N., and R.S.B. Green. 1979. The effect of aggregate size on the infiltration behaviour of a slaking soil and its relevance to ponded irrigation. Australian Journal of Soil Research. 17: 65–73.

Cook, D.F., and S.D. Nelson. 1986. Effect of polyacrylamide on seedling emergence in crust-forming soils. Soil Scence. 141:328-333.

Corey, A.T. 1977. Mechanics of heterogeneous fluids in porous media. Water Resources Publications.Ft. Collins, CO.

Davidson, R.A., C.F. Davidson, and A. Roa-Espinosa. 2009. Linear anionic polyacrylamide as an effective post-fire soil treatment: Understanding the chemistry and physical science. Journal of soil and water conservation. 64(4): 243-252.

Day, P.R. 1956. Report of the Committee on Physical Analyses, 1954–1955. Soil Science Society of America Proceedings. 20:167–169.

DeBano, L.F. 1981. Water repellent soils: A state-of-the-art. Gen. Tech. Rep. PSW-46. U.S. For. Serv., Pac. Southw. For. Range Exp. Stn., Berkeley, CA.

DeBano, L. F., D.G. Neary, and P. F. Folliott. 1998. *Fire's effect on ecosystems*. Wiley, New York.

DeBano, L.F .2000. The role of fire and soil heating on water repellence in wildland environments: a review. Journal of Hydrology. 231:195–206.

DeBano, L.F., D.G. Neary, and P.F. Ffolliott. 2005. Soil physical properties. p.29–51. In D.G. Neary et al. (ed.) Wildland fi re in ecosystems: Effects of fire on soil and water. Gen. Tech. Rep. RMRS-GTR-42. Vol. 4. U.S. For. Serv., Rocky Mountain Res. Stn., Fort Collins, CO.

Doerr, S.H., R.A. Shakesby, and R.P.D. Walsh. 2000. Soil water repellence: its causes, characteristics and hydrogeomorphological significance. Earth-Sci Rev. 51:33–65.

Doerr, S.H., and J.A. Moody. 2004. Hydrological impacts of soil water repellency: On spatial and temporal uncertainties. Hydrol. Processes 18:829–832.

Doerr S.H., S.W. Woods , D.A. Martin, and M. Casimiro. 2009a. 'Natural background' soil water repellency in conifer forests of the north-western USA: Its prediction and relationship to wildfire occurrence. Journal of Hydrology. 371:12–21.

Doerr, S.H., R.A. Shakesby, and L.H. MacDonald. 2009b. In A. Cerda and P.R. Robichaud (ed.) Fire effects on soils and restoration strategies. Science Publ., Enfield, NH.

Emerson, W.W. 1977. Physical properties and structure. In 'Soil factors in crop production in a semi-arid environment'. (Eds JS Russell, EL Greacen) pp. 78–104. (University of Queensland Press: Brisbane, Qld)

Flanagan, D.C., K. Chaudhari, and L.D. Norton. 2002. Polyacrylamide soil amendment effects on runoff and sediment yield on steep slopes: Part II. Natural Rainfall Conditions. Transactions of the ASAE .45(5): 1339-1351.

Frenkel, H., M.V. Fey, and G.J. Levy. 1992. Organic and inorganic anion effects on reference and soil clay critical flocculation concentration. Soil Science Society of America Journal 56:1762–1766.

Garcia-Corona, R., E. Benito, E. de-Blas, and M. E. Varela.2004. Effects of Heating on some Soil Physical Properties Related to its Hydrological Behavior in Two North-Western Spanish Soils. International Journal of Wildland fire .13(2): 195–199.

Gimeno-Garcia, E., V. Andreu, and J. L Rubio.2000. Changes in organic matter, nitrogen, phosphoros and cations in soil as a result of fire and water erosion in a Mediterranean landscape. European Journal of Soil Science. 51:201-210.

Ginsburg, P. 2000. Afforestation in Israel: A source of social goods and services, Journal of forestry. 98(3):32-36.

Giovannini, G., S. Luchéis, and M. Giachetti. 1988. Effect of heating on some physical and chemical parameters related to soil aggregation and erodibility. Soil Science. 146:255–262.

Giovannini, G., S. Luchéis, and M. Giachetti. 1990. Effects of Heating on some chemical parameters related to soil fertility and plant growth. Soil Sciance.149:344-350.

Giovannini, G., and S. Lucchesi. 1997. Modifications induced in soil physico-chemical parameters by experimental fires at different intensities. Soil Sci .162:479–486.

Giovannini, G., V.R. Vallejo, S. Lucchesi, S. Bautista, S. Ciompi, and J. Llovet . 2001. Effects of land use and eventual fire on soil erodibility in dry Mediterranean conditions. Forest Ecology and Management. 147: 15–23.

Gonza´lez-Perez ,J.A., F.J. Gonzalez-Vila, G. Almendros, and H. Knicker. 2004. The effect of fire on soil organic matter-a review. Environment International. 30:855– 870.

Green, V.S., D.E. Stott, J.G. Graveel, and L.D. Norton. 2004. Stability analysis of soil aggregates treated with anionic polyacrylamides of different molecular formulations. Soil Sci. 169:573–581

Greenland, D.J. 1972. Interactions between organic polymers and inorganic soil particles. In 'Proceedings of a Symposium on Fundamentals of Soil Conditioning'. (Ed.MDeBoodt) pp. 205–220. (State Univ. Ghent: Ghent, Belgium)

Gu, B., and H.E. Doner. 1992. The interaction of polysaccharides with Silver Hill illite. Clays and Clay Minerals. 40:151–156.

Gupta, R.K., D.K. Bhumbla, and I.P. Abrol . 1984. Effect of sodicity, pH, organic matter, and calcium carbonate on the dispersion behaviour of soils. Soil Science. 137: 245–251.

Helalia, A.M., and J. Letey. 1988. Polymer type and water quality effects on soil dispersion. Soil Science Society of America Journal .52: 243–246.

Hernandez, T., C. Garcıa, and I. Reinhardt.1997. Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. Biol Fertil Soils. 25:109–116.

Horne, D.J., and J.C. McIntosh. 2000. Hydrophobic compounds in sands in New Zealand-extraction, characterization and proposed mechanisms for repellence expression. J Hydrol. 231:35–46.

Huffman, E.L., L.H. MacDonald, and J.D. Stednick. 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrol Process. 15:2877–2892.

IFFN No 15. – September 1996, p:2-6. Forests and Forest Fires in Israel. http://www.fire.uni-freiburg.de/iffn/country/il/il_3.htm

Iglesias, T., V. Cala, and J. Gonzalez.1997. Mineralogical and chemical modifications in soils affected by a forest fire in the Mediterranean area. The Science of the Total Environment 204 :89-96.

Inbar, M., L. Wittenberg, and M. Tamir. 1997. Soil Erosion and Forestry Management After Wildfire in a Mediterranean Woodland, Mt. Carmel, Israel . International Journal of Wildland Fire .**7**: 285–294.

Inbar, M., M., Tamir. and L. Wittenberg. 1998. Runoff and erosion processes after a forest fire in Mount Carmel, a Mediterranean area. Geomorphology 24, 17–33.

Johansen, M.P., T.E. Hakonson, and D.D. Breshears. 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. Hydrol. Process. 15:2953–2965.

Johnes, S.M., T. Cummins, G.M. Boyel, J. Aherne, and E.P. Farrell. 1998. A pilot study into the effects of clearfelling on nutrient losses and sustainability. Monstac Project Final Report. Forest Ecosystem Research Group Rep 23, University College, Dublin, 62 pp

Kay, B.P., and D.A.Angers. 1999. Soil structure. In 'Handbook of soil science'. (Ed. ME Sumner) pp. A-229–A-269. (CRC Press: New York)

Keeley, J. E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. International Journal of Wildland Fire. 18:116–126

Kutiel, P., and Z. Naveh.1987. The effect of fire on nutrients in a pine forest soil. Plant and Soil. 104: 269-274 .

Kutiel , P., and A. Shaviv. 1989. Changes in soil N-P status in laboratory simulated forest fire. Plant and Soil. 120:56-63.

Kutiel, P., and A. Shaviv. 1992. Effects of soil types, plant composition and leaching on soil nutrients following a simulated forest fire. Forest Ecology and Management. 53: 329-343.

Kutiel, P., and M. Inbar. 1993. Fire impact on soil nutrients and soil erosion in a Mediterranean pine forest plantation. Catena . 20:129-139.

Kutiel, P., H. Lavee, M. Segev, and Y. Benyamini. 1995. The effect of fire-induced surface heterogeneity on rainfall-runoff-erosion relationships in an eastern Mediterranean ecosystem, Israel Catena. 25: 77-87.

Lado, M., Ben-Hur, and I. Shainberg. 2004a. Soil Wetting and Texture Effects on Aggregate Stability, Seal Formation, and Erosion. Soil Science Society of America Journal. 68:1992–1999.

Lado, M., A. Paz, and M. Ben-Hur. 2004b. Organic Matter and Aggregate Size Interactions in Infiltration, Seal Formation, and Soil Loss. Soil Science Society of America Journal. 68:935–942.

Lado, M., and M. Ben-Hur. 2009. Treated domestic sewage irrigation effects on soil hydraulic properties in arid and semiarid zones: a review. Soil & Tillage Research. 106:152-163.

Lavee, H., P. Kutiel, M. Segev, and Y. Benyamini. 1995. Effect of surface roughness on runoff and erosion in a Mediterranean ecosystem: The role of fire. Geomorphology. 11:227–234.

Larsen, I.J., L.H. MacDonald, E. Brown, D. Rough, M.J. Welsh, J. Pietraszek, Z. Libohova, and J.D. Benavides-Solorio, 2009. Causes of post-fire runoff and erosion: soil water repellency, cover or soil sealing? Soil Science Society of America Journal 73 (4): 1303–1407.

Lentz, R.D., and D.T. Westermann. 2010. Managing Runoff Water Quality from Recently Manured, Furrow-Irrigated Fields. Soil Science Society of America Journal. 74(4):1310-1319.

Lentz, R.D., and R.E. Sojka. 2009. Long-Term Polyacrylamide Formulation Effects on Soil Erosion, Water Infiltration, and Yields of Furrow-Irrigated Crops. Agronomy Journal. 101(2): 305-314.

Letey, J.F. 1969. Measurement of contact angle, water drop penetration time, and critical surface tension. p. 43–47. In L.F. DeBano and J.F. Letey (ed.) Proc. Symp. on Water Repellent Soils, Riverside, CA. May 1968. Univ. of California, Riverside.

Levy, G.J., M. Ben-Hur, and M. Agassi. 1991. The effect of polyacrylamide on runoff, erosion and cotton yield from fields irrigated with moving sprinkler systems. Irrigation Science. 12:55–60.

Levy, G.J., J. Levin, M. Gal, M. Ben-Hur and I. Shainberg. 1992. Polymers' Effects on Infiltration and Soil Erosion during Consecutive Simulated Sprinkler Irrigations. Soil Science Society of America Journal . 56 (3): 902-907.

Levy, G.J., and M. Ben-Hur. 1998. Some uses of water-soluble polymers in soil. Books in Soils, Plants, and the Environment. Handbook of soil conditioners. P: 399-428 .**Publisher:** Marcel Dekker, Inc., 270 Madison Avenue, New York, New York 10016, USA; Marcel Dekker, Inc., Basel, Switzerland.

Levy, G.J., and W.P. Miller.1999. Polyacrylamide adsorption and aggregate stability. Soil and Tillage Research. 51:121-128.

Ludwig, B., P.K. Khanna, R.J. Raison, K.L. Jacobsen. 1998. Modelling cation composition of soil extracts under ashbeds following an intense slashfire in a eucalypt forest. Forest Ecology and Management. 103:9–20.

Main Report, FAO Forestry Paper 163, Global Forest Resources Assessment, 2010, Food and Agriculture Organization of the United States, Rome, 2010, page Xiii

Malik, M., and J. Letey. 1991. Adsorption of polyacrylamide and polysaccharide polymers on soil materials. Soil Science Society of America Journal .55:380–383.

Malik, M., and J. Letey. 1992. Pore-sized-dependent apparent viscosity for organic solutes in saturated porous media. Soil Science Society of America Journal .56:1032–1035.

Mallik, A.U., C.H. Gimingham, and A.A. Rahman. 1984. Ecological ef ects of heather burning : I. Water ini ltration, moisture retention, and porosity of surface soil. Journal of Ecololgy. 72:767–776.

Mamedov, A.I., S. Beckmann, C. Huang, and G.J Levy. 2007. Aggregate Stability as Affected by Polyacrylamide Molecular Weight, Soil Texture, and Water Quality. Soil Science Society of America journal. 71(6): 1909-1918.

Martin, D.A., and J.A. Moody. 2001. Comparison of soil infiltration rates in burned and unburned mountainous watersheds. Hydrological Processes. 15:2893–2903.

Mataix-Solera , J., V. Arcenegui , C. Guerrero , M.M. Jordán , P. Dlapa , N. Tessler , and L. Wittenberg. 2008. Can *terra rossa* become water repellent by burning? A laboratory approach. Geoderma .147: 178–184.

Mataix-Solera, J., and S.H. Doerr. 2004. Hydrophobicity and aggregates stability in calcareous topsoils from fire-affected pine forest in southeastern Spain. Geoderma .118: 77–88.

McIntyre, D.S. 1958. Permeability measurements of soil crust formed by raindrop impact. Soil Sci. 85:158–189.

Meyer, L.D., and W.C. Harmon. 1984. Susceptibility of agricultural soils to interrill erosion. Soil Sci. Soc. Am. J. 48: 1152-1157.

Michaels, A.S. 1954. Aggregation of suspensions by polyelectrolytes. Ind. Eng. Chem. 46(7): 1485–1490.

Moody, J.A., and D.A. Martin. 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surf. Processes Landforms. 26:1049–1070.

Morin. J., D. Goldberg, and I. Seginer. 1967. A rainfall simulator with a rotating disk. Trans. AEAE. 10:74-77,79.

Morin, J., Y. Benyamini, and A. Michaeli. 1981. The effect of raindrop impact on the dynamics of soil surface crusting and water movement in the profile. Journal of Hydrology. 52:321-335.

Nadler, A., M. Malik, and J. Letey. 1992. Desorption of Polyaacrylamide and Polysaccharide Polymers from Soil Meterials. Soil Tecnlogy. 5: 91-95.

Neary, D.G., C.C Kloptek, L.F DeBano, and P.F Ffolliott.1999. Fire effect on belowgrownd sustainability: a review and synthesis. Forest Ecology and Management .122: 51-71.

Neary, D.G., K,C. Ryan, L.F. DeBano. 2005 (Ed), Wildland fire in ecosystems: effects of fire on soil and water. USDA Forest Service, Rockey Mountains Research Station, General Technical report RMRS-GTR-42-volume 4. P:62.

O'Brien, N.R. 1971. Fabric of kaolinite and illite floccules. Clays and Clay Minerals .19:353–359.

Onda, Y., W.E. Dietrich, and F. Booker. 2008. Evolution of overland flow after a severe forest fire, Point Reyes, California. Catena 72:13–20.

Osem, Y., P. Ginsberg, I. Tauber, N. Atzmon, and A. Perevolotsky.2008. Sustainable Management of Mediterranean planted coniferous Forests: An Israeli Definition. Journal of forestry. 106 (1): 38-46.

Pardini, G., M. Gispert, and G. Dunjo. 2004. Relative influence of wildfire on soil properties and erosion processes in different Mediterranean environments in NE Spain. Science of the Total Environment. 328:237–246.

Pausas .J G., Vallejo V.R. 1999. The role of fire in European Mediterranean Ecosystems .ChuviecoE(ed)Remote sensing of large wildfires in the European Mediterranean Basin. Springer,Berlin, pp3–16.

Pausas, J.G., N. Ouadah, A. Ferran, T. Gimeno and R.Vallejo. 2003. Fire severity and seedling establishment in *Pinus halepensis* woodlands, eastern Iberian Peninsula. Plant Ecology. 169: 205–213.

Pausas, J.G. 2004. Changes in fire and climate in the eastern Iberian peninsula (Mediterranean basin), Climatic Change. 63(3): 337–350.

Pausas, J.G and Fernández-Muñoz, S. 2009. Fire regime changes in the Western Mediterranean Basin: from fuel-limited to drought-driven fire regime. Climatic Change. 110(1-2): 215-226.

Peng, F.F., P. Di, 1994. Effect of Multivalent Salts-Calcium and Aluminum on the Flocculation of Kaolin Suspention with Anionic Polyacrylamide. Journal of Colloid Interface Science. 164:229–237

Prosser, I.P., and L. Williams. 1998. The effect of wildfire on runoff and erosion in native eucalyptus forest. Hydrol. Processes. 12:251–265.\

Quirk, J.p. and R. K. Schofield. 1955. The effect of electrolyte concentration on soil permeability. Journal of Soil Science. 6(2):163-178.

Rengasamy, P. and K. A. Olsson.1991. Sodicity and soil structure. Australian Journal of Soil Research. 29(6):935 – 952.

Rapp, I., I. Shainberg, and A. Banin. 2000. Evaporation and Crust Impedence Role in Seedling Emergence. Soil Science. 165(4) : 354-364.

Renard, K.G., G.R. Foster, G.A. Weesies, D.K. McCool, and D.C. Yoder. 1997. Predicting soil erosion by water: A guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agric. Handbk. 703. USDA, Washington, DC

Roberts K, Kowalewska J, Friberg S . 1974. The influence of interactions between hydrolyzed aluminum ions and polyacrylamides on the sedimentation of kaolin suspensions. Journal of Colloid and Interface Science .48:361–367

Robichaud, P.R., and R.E. Brown. 1999. What happened after the smoke cleared: Onsite erosion rates after a wildfire in eastern Oregon. p. 419–426. In D. Olsen and J.P. Potyondy (ed.) Proc. AWRA Specialty Conf. on Wildland Hydrol., Bozeman, MT. 30 June–2 July 1999. Am. Water Resour. Assoc., Herndon, VA.

Robichaud, P.R. 2000. Fire effects on infiltration rates after prescribed fire in northern Rocky Mountain forests, USA. J. Hydrol. 231:220–229.

Robichaud, P.R. 2005. Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery. International Journal of Wildland Fire .14: 475-485.

Robichaud, P.R., L.E. Ashmun, and B.D. Sims. 2010. Post-Fire Treatment Effectiveness for Hillslope Stabilization. United States Department of Agriculture. Forest Service. Rocky Mountain Research Station. General Technical Report RMRS-GTR-240.

Rough, D. 2007. Effectiveness of rehabilitation treatments in reducing post-fire erosion after the Hayman and Schoonover Fires, Colorado Front Range. Fort Collins, CO: Colorado State University. 186 p. Thesis.

Rubio, H.O., M.K. Wood, M. Cardenas, and B.A. Buchanan. 1989. Effect of polyacrylamide on seedling emergence of three grass species. Soil Science .148(5): 355–360.

Santos, F.L., J. L. Reis, O.C. Martins, N.L. Castanheira, and R.P. Serralheiro. 2003. Comparative Assessment of Infiltration, Runoff and Erosion of Sprinkler Irrigated Soils. Biosystems Engineering .86 (3): 355–364.

Sato, T., and R. Ruch. 1980. 'Stabilization of colloidal dispersions by polymeric adsorption.' (Marcel Dekker: New York)

Schamp, N., and J. Huylebroeck. 1973. Adsorption of polymers on clays. Journal of Polymer Science: Polymer Symposia. 42(2):553-562.

Schamp, N., J. Huylebroeck, and M. Sadones. 1975. Adhesion and adsorption phenomena in soil conditioning. In 'Soil conditioners'. Soil Science Society of America Special Publication. 7: 13–22. (American Society of Agronomy: Madison, WI)

Seybold, C.A. 1994. Polyacrylamide review: soil conditioning and environmental fate. Comm. Soil Sci. Plant Anal. 25: 2171–2185.

Shainberg, I., and Letey, J. 1984. Response of soils to sodic and acidic conditions. Hilgadia. 52: 1-57.

Shainberg, I., D.N. Warrington, and P. Rengasamy. 1990. Water quality and PAM interactions in reducing surface sealing. Soil Science. 149:301–307.

Shainberg, I., G.J. Levy, P. Rengasamy, and H. Frenkel. 1992. Aggregate stability and seal formation as affected by drops' impact energy and soil amendments. Soil Sci. 154: 113-119.

Shainberg, I., and G.J. Levy. 1994. Organic polymers and soil sealing in cultivated soil. Soil Science. 158:267–273.

Shainberg, I., and G.J. Levy. 1996. Infiltration and seal formation processes. p. 1–22. In M. Agassi (ed.) Soil erosion, conservation, and rehabilitation. Marcel Dekker, New York.

Shakesby, R.A., S.H. Doerr, and R.P.D. Walsh. 2000. The erosional impact of soil hydrophobicity: Current problems and future research directions. J. Hydrol. 231:178–191.

Shakesby, R. A., and S. H. Doerr. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews. 74: 269-307.

Shakesby, R.A., 2011. Post-wildfire soil erosion in the Mediterranean: Review and future research directions. Earth-Science Reviews. 105: 71-100.

Singer, A. 2007. The soils of Israel. Springer-Verlag, Berlin.

Smith, H.J.C., G.J. Levy, and I. Shainberg. 1990. Water-droplet energy and soil amendments: Effect on infiltration and erosion. Soil Science Society of America Journal .54:1084–1087.

soil. Soil Sci Soc Am J 51:1299–1302.

Sojka, R.E., D.L. Bjorneberg, J.A. Entry, R.D. Lentz, and W.J. Orts.2007. Polyacrylamide in agriculture and environmental land management. Advances in Agronomy. 92:75-162.

Steel R. G. D, Torrie, J.H. 1981. Principles and procedures of statistics: a biometrical approach. McGraw-Hill: London.

Steinberger, Y., and N.E West.1991. Effects of Polyacrylamine on some biological and physical features of soil: preliminary results from a greenhouse study. Arid Soil Research and Rehabilitation. 5:77-81.

Stern, R., A.J. Van Der Merwe, M.C. Laker, and I. Shainberg. 1992. Effect of soil surface treatments on runoff and wheat yields under irrigation. Agronomy. Journal. 84:114-119.

surface crusting and water movement in the profile. J Hydrol. 52:321–335.

Tang, Z., T. Le, J. Yu, I. Shainberg, A.I. Mamedov, M. Ben-Hur, and G.J. Levy. 2006. Runoff and interrill erosion in sodic soils treated with dry PAM and phosphogypsum. Soil Science Society of America Journal. 70: 679-690.

Tarchitzky, J., Y. Golobati, R. Keren, and Y. Chen. 1999. Wastewater effects on montmorillonite suspensions and hydraulic properties of sandy soil. Soil Sci. Soc. Am. J. 63:554–560.

Terefe, T., I. Mariscal-Sancho, F. Peregrina, and R. Espejo. 2008. Influence of heating on various properties of six Mediterranean soils: A laboratory study. Geoderma. 143:273–280.

Theng, B.K.G. 1982. Clay-polymer interactions: summary and prospectives. Clays and Clay Minerals .30:1–10.

Tisdall, J.M., and J.M Oades. 1982. Organic Matter and Water-Stable Aggregates In Soils. Joural of soil science. 33:141-163.

Trout, T.J., R.E Sojka, and R.D Lentz. 1995. Polyacrylamide Effect on Furrow Erosion and Infiltration. American Society of Agricultural Engineers. 38(3):761-765.

Úbeda, X., P. Pereira, L. Outeiro, and D.A. Martin.2009. Effects of fire temperature on the physical and chemical characteristics of the ash from two plots of cork oak (Quercus suber). Land Degradation & Development. 20(6): 589–608.

Ulery, A.L., and R.C. Graham. 1993. Forest fire effects on soil color and texture. Soil Science Society of America Journal. 57:135–140.

Van Dijk, A.I.J.M., and R.J. Keenan. 2007. Planted forests and water in perspective. For Ecol Manage. 251:1–9.

Van Olphen, H. 1977. An introduction to clay colloid chemistry. 2nd edn (Interscience Publishing: New York)

Varela, M.E., E. Benito, and J.J. Keizer. 2010. Effects of wildfire and laboratory heating on soil aggregate stability of pine forests in Galicia: The role of lithology, soil organic matter content and water repellency. Catena. 83: 127–134.

Vélez, R.1990. Mediterranean forest fires: A regional perspective, Unasylva .162: 10–12.

Vélez, R. 2002. Causes of forest fires in the Mediterranean Basin. EFI Proceedings. 45: 35-42.

Volk, H., and R.E. Friedrich. 1980. Polyacrylamide. In 'Handbook of water soluble gums and resins'.(Ed. RL Davidson) pp.16-1–16-26 (McGraw Hill: New York)

Wallace, A. 1987. Anionic polyacrylamide treatment of soil improves seedling emergence and growth. HortScience. 22(5): 951.

Witenberg, L., and D. Malkinson. 2009. Spatio-temporal perspectives of forest fires regimes in a maturing Mediterranean mixed pine landscape. Eur J Forest Res .128:297–304.

Wittenberg, L., and M. Inbar. 2009. The Role of Fire Disturbance on Runoff and Erosion Processes – a Long-Term Approach, Mt. Carmel Case Study, Israel. Geographical Research. 47(1):46–56.

Woods, S.W., and V.N. Balfour. 2010. The effects of soil texture and ash thickness on the post-fire hydrological response from ash-covered soils. Journal of Hydrology. 393:274-286.

Yu, J., T. Lei, I. Shainberg, A.I. Mamedov, and G.J. Levy. 2003. Infiltration and erosion in soils treated with dry PAM and gypsum. Soil Science Society of America Journal .67:630–636.

8 תקציר

שריפות יער הן בעיה כלל עולמית ככלל, ובאגן הים התיכון בפרט, כשתנאי האקלים מאופיינים בקיץ ארוך ויבש וחורף גשום, מאפשרים התפתחות של תנאי שריפה, במיוחד בתקופה של שינויי אקלים, עלייה בכמות האוכלוסייה העולמית ושינויים בשימושי קרקע. שריפות גורמות לשינויים סביבתיים בעיקר ע״י פגיעה בצומח המקומי והשפעתן על הקרקע, אשר יכולים להשפיע על המעגל ההידרולוגי האזורי. שריפות שונות אחת מהשנייה בחומרתן (severity) ואף בתוך אותה שריפה עלולים להיווצר תנאי חומרה שונים, התלויים בתנאי האקלים והרוח בזמן השריפה, הטופוגרפיה והתפרוסת המרחבית של חומרי בעירה. חומרות השריפה השונות יוצרות כיסי השפעה שונים במרחב על הקרקע שמתחתיהן. אחת מהההשפעות הבולטות של שריפות יער היא עלייה בנגר ובסחף קרקע טרם שיקום הצמחיה המקומית. פוליאקרילאמיד אניוני (PAM) הוכח בחקלאות כמייצב מבנה קרקע יעיל העוזר להקטין נגר וסחף ובייצוב מדרונות תלולים. ה-PAM זול ומיושם בכמויות קטנות יחסית על מנת לקבל תוצאות חיוביות. יעילותו של ה-PAM כמייצב קרקע בעקבות שריפה נבדקה אך במעט, ויעילותו מעולם לא נבדקה על קרקע שנחשפה לחומרות שריפה שונות. מטרות המחקר היו: (א) בחינת השפעת תנאי שריפה שונים על התכונות הכימיות והכימיות-פיזיקאליות בקרקע רנדזינה בהירה, והשפעתם על יחסי חידור-נגר עילי-וסחף קרקע; (ב) בחינת השפעת טיפול ב-PAM לאחר שריפות בחומרות שונות על סחף קרקע, בתנאים מבוקרים ובניסוי שדה פרילימינרי; (ג) בחינת התחלתית שתי כמויות של PAM על שיקום צמחיה באמצעות חיטה תרבותית (.Triticum aestivum L) כצמח מודל לשיקום צמחייה עשבונית חד שנתית.

על מנת לבדוק את השפעת תנאי השריפה על התכונות הכימיות והכימיות-פיזיקאליות בקרקע השתמשנו בקרקע רנדזינה בהירה מיער בירייה אשר נחשפה לתנאי שריפה שונים המייצגים את ההטרוגניות המרחבית שיכולה להיות בשטח שנחשף לשריפת יער, והשפעתה על הקרקע. קרקע לא שרופה (UB), קרקע יער שנחשפה לשריפה ישירה בעצמה קלה-בינונית (DF) וקרקע שנחשפה לחום בינוני לאורך זמן, אך לא נחשפה לשריפה ישירה (HT) נבחרו לייצג את המצבים שלהלן, בהתאמה: קרקע מאזור שלא הושפע מהשריפה, קרקע שהושפעה ישירות משריפה בעצמה נמוכה-בינונית; וקרקע שהושפעה מטמפרטורה אך לא לאש.. קרקע רנדזינה בהירה נדגמה מחוץ ומתוך (UB ו-DF, בהתאמה) אזור שרוף שנשרף בעצמה נמוכה-בינונית ביער בירייה. חצייה של הקרקע הלא שרופה נשרף בתנור ב- 300°c ל-8 שעות (HT). התוצאות הראו שקרקע שנחשפה לחום (HT) עברה שינויים פיזיקליים וכימיים-פיזיקליים בתכונותיה בהשוואה לקרקע לא שרופה (UB) ולקרקע שנשרפה בשריפת ישירה (DF). בין השינויים שנמצאו היו ירידה בכמות החומר האורגני (ח״א), ה-SAR וה-CEC ובפרקציה החרסיתית; ועלייה בתכולת הגיר, ובח״א המסיס והמוליכות החשמלית (EC) של תמיסת הקרקע. קרקע DF לא היתה שונה ברוב התכונות מקרקע-UB פרט לעלייה בתכולת ח״א וב-CEC, וירידה ב-SAR.

ניסויים במדמה גשם של שלוש סופות עוקבות של 80 מ״מ כ״א בעצמה של 47 מ״מ/שעה, עם תקופת ייבוש של 72 שעות בין סופה אחת לשנייה, נערכו עם שלושת הקרקעות הנ״ל. שוני בערכי החידור, בכמות הנגר העילי והסחף נמצאו בין שלושת הקרקעות. דחיית מים נמוכה נמצאה בכל שלושת הקרקעות, ללא הבדל

בינהן, ולכן נפסלה האפשרות ששוני בעצמת דחיית המים גרם להבדלים בערכי החידור בין הקרקעות. למרות הירידה בחⅠⅠא, לקרקע HT היו ערכי החידור הגבוהים ביותר, וכמויות נגר עילי וסחף הקטנים ביותר. הסיבות לתוצאות הנⅠⅠל הוערכו ביציבות גדולה יותר במבנה הקרקע בעקבות התלכדות של חלקיקי חרסית בתלכידים כתוצאה מהטמפרטורה הגבוהה, ובעקבות שינויים כימים-פיזיקליים בתמיסת הקרקע שמנעו דיספרסיה של חרסיות. התלכדותם של חלקיקי החרסית גרם לעליה ביציבות תלכידי הקרקע שהוכחה עⅠⅠי ביצוע בדיקות מיגוג (slaking). כמו כן, עⅠⅠי מבחני דיספרסיה, נמצא שהדיספרסיה בקרקע זו היתה נמוכה מהאחרות, כנראה בגלל עליה בⅠEC וירידה בⅠSAR בתמיסת הקרקע, ובנוסף להתלכדות החרסיות גרמו לייצוב מבני. למרות שלקרקע DF ערכי חידור נמוכים יותר מקרקע HT, הם היו גבוהים יותר מקרקע UB ולכן נפח הנגר העילי והסחף בקרקע DF היו נמוכים יותר בסופה הראשונה והשניה. לעומת זאת, במבחני מיגוג ודיספרסיה לא נמצאו הבדלים מובהקים בין קרקע DF לקרקע UB, אם כי ערך הדיספרסיה שלקרקע DF היה נמוך יותר, כנראה כתוצאה מערכי SAR נמוכים, עובדה שיכולה להסביר את ההבדלים בסחף בין שתי הקרקעות הנⅠⅠל. חוסר ההבדל ביציבות מוכיח שכמות החⅠⅠא הגבוהה יותר בקרקע DF לא לקחה חלק בייצוב תלכידי הקרקע. הסיבה המשוערת לעליה זו בחⅠⅠא היא תוספת של חⅠⅠא ממקור צימחי או מהנשר שלא עבר מינרליזציה מלאה בזמן השריפה וערבב עם הקרקע בזמן הדיגום. בדומה לקרקע HT, ההבדלים בערכי החידור קטנו בסופה השנייה והשלישית, והגיעו לערכים דומים בסוף הסופה האחרונה. המסקנה שהוופקה היא שקרקע רנדזינה בהירה שנחשפה לתנאי שריפה שונים מתייצבת עⅠⅠי שינויים כימיים וכימיים-פיזיקליים בקרקע ובתמיסת הקרקע, אשר מעלים את כוחות ההתנגדות של התלכידים למכות טיפות הגשם, להרטבה מהירה הגורמת למיגוג, לדיספרסיה של חרסיות ובעקבות כך ליצירת קרום על פני הקרקע. לעומת זאת, מהעובדה שההבדלים בערכי החידור הולכים ומצטמצמים בסופות העוקבות, אנו מסיקים שיציבות הקרקע לא נשמרת לאורך זמן, במיוחד אם הקרקע נשארת חשופה בעקבות פגיעה בצמחיה כתוצאה מהשריפה. ולכן, אני מציעים שימוש באחד מאמצעי מניעת סחף ככלי ממשקי לטיפול בנזקי השריפה.

בניסויי מדמה גשם דומים עם אותן קרקעות עם תוספת PAM גבישי בכמות של 50 קⅠⅠג/הקטר נמצא שבסופה הראשונה ערכי החידור בקרקעות DF וⅠUB ירדו בצורה מובהקת וגרמו לעליה בנגר כתוצאה לעלייה בצמיגותו, אך לעומת זאת נצפתה ירידה בכמות הסחף. בסופות העוקבות, טיפול הⅠPAM הקטין את כמות סחף בהשוואה לביקורת, והגדיל את ערכי החידור והקטין את הנגר העילי בהשוואה לסופה הראשונה. אנו מניחים שהפולימר מקטין את הסחף במנגנון בעל שני שלבים : (א) העלייה בצמיגות הנגר העילי אמנם גורמת לירידה בערכי החידור ולעלייה בסחף, אך היא גם גורמת לו לנוע באיטיות על פני הקרקע ובכך קטנה יכולת ההרס שלו וכושר הנשיאה שלו. בנוסף, כושר הפלוקולציה (הפתחה) של מולקולות הפולימר גורמות ליצירת פתיתים במי הנגר העילי ומקטינה את ריכוז הסדמנטים שבו ובכך מקטינה עוד את כושר הנשיאה שלו.(ב) לאחר תקופת הייבוש שבין הסופות, מולקולות הפולימר שלא נשטפו עם מי הנגר נספחות לחלקיקי הקרקע על פני השטח ומייצבות אותם. מנגנון ייצוב זה גורם לעלייה בערכי החידור ולירידה בנפח הנגר העילי ובכמות הסחף בסופות העוקבות. מנגנון הייצוב האחרון הוכח עⅠⅠי מבחני מיגוג ודיספרסיה בנוכחות פולימר. בניגוד לשתי הקרקעות האחרות, קרקע הⅠHT ביטאה דפוס התנהגות אחר מאחר ולא נצפתה ירידה בסחף במהלך

הסופה הראשונה. הסיבה לתוצאה הנ"ל מוערכת בעלייה ב-EC של הנגר הגורמת לירידה בצמיגותו ולעלייה בתכונות ההרס שלו ולעלייה בסחף. אך מצד שני הצמיגות גבוהה יחסית בכדי לאפשר ערכי חידור גבוהים ולכן נפח הנגר עולה. בסופות העוקבות ערכי החידור עולים ונפח הנגר וכמות הסחף יורדים כתוצאה ממנגנון הייצוב. בחינת אותו טיפול PAM על קרקע יער (non calcareous kaolinitic sandy loam מברבנסה שבגליסיה, ספרד) בעלת הרכב מכאני ומינרולוגיה שונים שנחשפה לטיפול של חום, גרם לירידה בכמות הסחף גם בסופה הראשונה בעקבות ערכי צמיגות יחסית גבוהים, וכך גם בסופות העוקבות.

על מנת לבדוק את ההשפעה של PAM בתנאי שדה, שנים עשר (12) חלקות נגר נבנו באזור השרוף המתואר לעיל, על מדרון מערבי אחיד של 40%. שלושה טיפולים עם ארבע חזרות כ"א נבדקו: ביקורת, 25 ו-50 ק"ג/הקטר של פולימר גבישי. מרבצי אפר שעל הקרקע הוסרו בעדינות על מנת לבדוק את השפעת הפולימר על הקרקע המינרלית בלבד. גבישי ה-PAM פוזרו ידנית מעט לאחר תחילת החורף של 2009-2010. לאחר סופות החורף השונות, סחף קרקע נאסף ממשפכים הממוקמים בתחתיתה של כל חלקת נגר, נמדד נפחו של הנגר בחביות האיסוף ודוגמאות ממנו נלקחו לקביעת ריכוז תרחיפי הסדימנטים וכימיה שבו. בנוסף, בכל אירוע גשם נמדדה כמותו של הגשם שבסופה ע"י חמישה מדי גשם זעירים שמוקמו במקום בתחילת החורף. נמצא יחס ליניארי ישר בין כמות הנגר המצטברת לבין כמות הגשם המצטברת שנמדדו במהלך החורף. אמנם לא בצורה מובהקת, טיפול ה-50 ק"ג/הקטר נמצא היעיל ביותר בהקטנת נגר וגרם להקטנת הסחף ב-50% ביחס לביקורת. לעומתו, טיפול ה-25 ק"ג/הקטר הקטין את הסחף ב-23% אך לא נמצא יעיל בהקטנת נפח הנגר. אמנם מנגנון הצמיגות שזוהה בניסויי מכשיר הגשם הקטין את הסחף בטיפולי הפולימר בסופה הראשונה, אך יעילותו של המנגנון הייצוב הוכחה כיציבה יותר, שכן גרמה להורדת כמות הסחף לאורכה של כל תקופת הגשמים. על מנת לבדוק את שיקום הצמחיה בעקבות טיפולי PAM, נבנו 15 חלקות של $0.25m^2$ סמוך לחלקות הנגר. 30 זרעים של חיטה תרבותית ($Triticum\ aestivum$ L) נזרעו במרכזה של כל חלקה, וטופלו באותם טיפולי PAM עם חמש חזרות לכל טיפול. לא נמצאו הבדלים מובהקים בין טיפולי הפולימר השונים והביקורת בביומסה על-קרקעית וביצירת זרעים לחלקה. בעקבות תוצאות אלו, אנו מניחים שטיפול ב-PAM בכמויות אלו לא ישפיעו על שקומה של הצמחיה העשבונית החד שנתית בעקבות השריפה.

שימוש בפולימרים סינטטיים להקטנת סחף קרקע לאחר שריפות יער

חיבור זה הוגש כעבודת גמר לקראת התואר "מוסמך אוניברסיטה"

על-ידי

אסף ענבר