RESEARCH ARTICLE

Effects of Post-Fire Vegetation Recovery on Soil Erosion in Vulnerable Montane Regions in a Monsoon Climate: A Decade of Monitoring

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Abstract

In areas with frequent fires and erosive soils under monsoon climate, we aimed to determine the long-term dynamics of soil loss during vegetation recovery and to examine elapsed time for soil stabilization. Runoff plots were installed in *Pinus densiflora* forests affected by stand-replacing fires on the East Coast of South Korea, which occurred in spring 2000. Runoff plots measured runoff and sediment yields for 11 years (2003–2013) during which time, vegetation cover of low-, intermediate-, and high-vegetation cover plots increased from 21 to 44%, 49% to 69%, and 87% to 95%, respectively. Vegetation was effective in preventing runoff and sediment yield. Nevertheless, to stabilize to below baseline load, it took 7 years at low and intermediate cover plots and only 3 years at high cover plots. For 7 years, 7.06 (5.2-fold) and 4.29 (3.1-fold) kg m^{-2} of soil were lost at the low and intermediate cover plots, respectively, compared to the high cover plots (1.37 kg m^{-2}). Sediment loss fluctuated more than runoff following extreme rainfall events. We suggest that for the slow recovery area (< 70% cover), appropriate measures should be introduced to prevent soil erosion immediately after a fire, and logging should be postponed until the soil is stabilized for 3 years even in fast recovery areas (> 70% cover).

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Introduction

The temperate monsoon climate to which Korea belongs is characterized by dryness in the spring and heavy rainfall in the summer (Korea Meteorological Administration 2019). Two-thirds of forest fires in Korea occur in the spring due to prolonged drought from fall to spring and strong winds, which occur in the spring as well (Choung 2002). Forest floors, where aboveground vegetation was removed by fire, are exposed to heavy rains in summer. More than 50% of annual precipitation falls in summer. Additionally, the occurrence of typhoons often overlaps with the monsoon period (Korea Meteorological Administration 2019).

Moreover, the terrain in Korea is mountainous with steep slopes where many of the forests have been devastated for a long time (Lee 2010; Choung et al. 2020). As a result, 69.2% of the soil is classified as Inceptisol and 13.7% as Entisol in South Korea (Rural Development Administration 2019). Inceptisol is defined as soils of relatively new origin and the weakest appearance of layers, and Entisol is less developed soil than the former. Weather regime, topography, heavy forest use, and disturbance can make soil more vulnerable to

erosion. In addition, fire not only removes vegetation, but also changes soil properties, such as permeability and porosity (Cerdá and Doerr 2005; Certini 2005; Durán Zuazo and Rodríguez Pleguezuelo 2008).

Due to the difficult climate conditions, vegetation recovery is particularly important to reduce the degree of soil erosion in South Korea. In fact, vegetation is well known as a key factor in the prevention of erosion based on its ability to intercept rainfall, reduce the impact of raindrops, reinforce soil with roots, and increase soil water infiltration (Cerdá and Doerr 2005; Badía and Martí 2008; Durán Zuazo and Rodríguez Pleguezuelo 2008; Shin et al. 2013; Wittenberg et al. 2014).

The East Coast fire, which occurred in 2000, damaged 23,794 ha of forests in the east coast region of South Korea, where large wildfires occur repeatedly and more frequently when compared to other regions (Choung 2002). Consequently, following the East Coast fire, the soil conditions became poorer and particularly erosive (Kim et al. 2008; Shin et al. 2013; Rural Development Administration 2019). To avoid recurring damages caused by forest fires, it is necessary to establish a comprehensive restoration plan including soil conservation.

The role of vegetation in suppressing general soil erosion has been well documented in wildfire areas (Certini 2005; Durán Zuazo and Rodríguez Pleguezuelo 2008). However, the pattern of soil erosion is site-specific as many complex factors including climate, topography, soil, and vegetation influence soil erosion. In the East Coast region, Lee et al. (2004) and Shin et al. (2008) investigated soil hydrology and erosion on hill slopes with similar conditions. However, vegetation was not considered a major variable and the investigation was conducted in a relatively short period. As vegetation develops and the soil stabilizes, the pattern of soil erosion will change over a long period of time; therefore, studies on long-term trends are necessary (Cerdá and Doerr 2005).

Rapid recovery of vegetation after fires has been reported in the East Coast region (Lee and Hong 1998; Choung et al. 2004; Choung and Choung 2019) though post-fire vegetation recovery can be spatially and temporally heterogeneous within the same area due to differences in fire severity, soil fertility, and the pre-fire forest structure and floristic composition (Choung et al. 2004; Shin et al. 2008). Differences in the speed of vegetation recovery lead to mosaic patterns of plant communities in the areas affected by fire (Turner et al. 1997).

In the present study, we aimed to determine the long-term dynamics of soil loss during vegetation recovery in monsoon climates where areas have unstable soils. We selected the most predominant *Pinus densiflora* forests affected by the stand-replacing fire, which occurred on the East Coast of Korea in the spring of 2000. We specifically considered

the following conditions for studied hillslopes: (i) weather regime, topography, and soil conditions were similar, but (ii) degrees of vegetation regeneration were different. Fourteen runoff plots were installed, and runoff and sediment yield were measured with vegetation development over a period of 11 years, between 2003 and 2013.

Results

Vegetation Recovery

In the low cover plots, mean vegetation cover rose from 21% in 2003 to 44% in 2010 (Fig. 1). Mean vegetation cover of intermediate cover plots also increased from 49% in 2003 to 69% in 2011. Mean vegetation cover of the high cover plots had reached 87% in 2003, and these levels were maintained between 85 and 95% throughout the study period. Due to logging in some runoff plots in October 2011, mean vegetation cover decreased in 2012 and 2013.

In 2003, covers of a Leguminosae shrub, *Lespedeza cyrtobotrya* (20.8%) and a Fagaceae tree, *Quercus mongolica* (19.8%) were greater than other plant species in the shrub layers in the high cover plots (Table 1). In the same layers at the low and intermediate cover plots, *L. cyrtobotrya*, *Prunus sargentii*, *Q. mongolica*, *Qurecus serrata*, and *Quercus variabilis* were noted; however, the cover of each species was less than 5.0%. In the herbaceous layers, *L. cyrtobotrya*, *Pteridium aquilinum* var. *latiusculum* and *Q. mongolica* were the dominant plant species.

In 2008, increases in the covers of *P. densiflora* and *Q. mongolica* were observed in the shrub layers (Table 1). Although *P. densiflora* was not observed in 2003, its cover
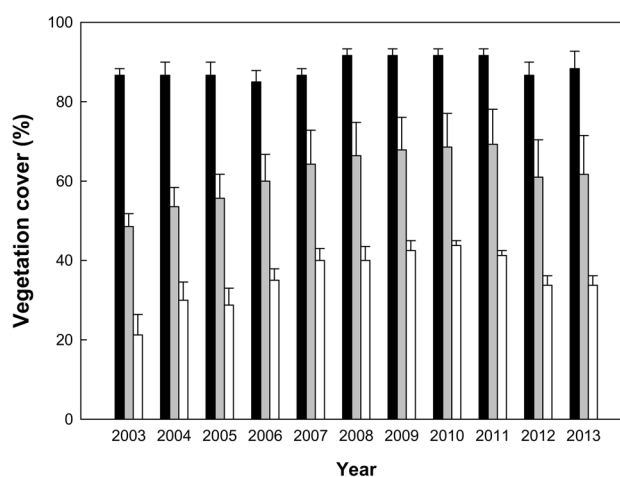

Fig. 1 Changes in vegetation cover (%) of runoff plots with high (black bar), intermediate (grey bar), and low (white bar) vegetation cover from 2003 to 2013, 3 to 13 years after the east coast forest fire in 2000. Data indicate means and standard error

Table 1 Mean cover (%) of shrub layers (> 0.5 m tall) and herbaceous layers measured at runoff plots with high, intermediate, and low vegetation cover in 2003 and 2008. Only species with > 0.5% vegetation cover are presented

Species	2003			2008		
	High	Intermediate	Low	High	Intermediate	Low
Shrub layer (> 0.5 m tall)						
Fraxinus sieboldiana	1.9					
Lespedeza cyrtobotrya	20.8	1.9	0.8	18.3	9.3	1.8
Pinus densiflora				11.7	17.1	
Prunus sargentii	2.0			10.0		
Quercus dentata	1.3		0.3			
Quercus mongolica	19.8	3.3	5.0	38.3	7.9	5.8
Quercus serrata	2.8	0.8			5.0	
Quercus variabilis			5.0			
Arundinella hirta	0.1	8.0	5.7		9.3	10.8
Carex lanceolata	3.3					
Chrysanthemum zawadskii subsp. *latilobum*	1.1	0.2				
Fraxinus sieboldiana	2.6	0.7	0.3	2.3	0.7	
Lespedeza cyrtobotrya	12.8	12.6	4.0		9.7	0.5
Lindera obtusiloba	1.4					
Lysimachia clethroides	5.3					
Miscanthus sinensis var. *purpurascens*	1.1					
Patrinia scabiosifolia	0.1	1.9				
Potentilla freyniana	1.3	0.1				
Pteridium aquilinum var. *latiusculum*	9.8	8.4	3.8	5.0	11.7	17.5
Quercus dentata	1.1	0.4				
Quercus mongolica	8.0	2.0	0.3	13.3	2.9	4.3
Quercus serrata		1.5	1.4	5.0	3.1	1.3
Quercus variabilis	1.4	0.1		6.7		
Rhododendron mucronulatum var. *ciliatum*	0.7	0.8				
Rhododendron schlippenbachii	0.8	2.4				
Rhus trichocarpa	2.1	0.5				
Smilax china	2.8					
Spodiopogon sibiricus	2.5	2.5	1.4			

in the shrub layers reached 11.7% and 17.1% in the high and intermediate cover plots, respectively. The cover of *Q. mongolica* increased from 19.8% and 3.3% in 2003 to 38.3% and 7.9% in the high and intermediate cover plots, respectively. These values did not change considerably in the shrub layers of low cover plots. The cover of *Q. serrata* at the shrub layers of intermediate cover plots increased from 0.8% in 2003 to 5.0% in 2008. Notably, *Quercus dentata* and *Q. variabilis*, which were both observed in 2003, were no longer present in 2008.

In herbaceous layers, the cover of *L. cyrtobotrya* decreased compared to 2003, and *L. cyrtobotrya* was not observed in 2008 in the high cover plots (Table 1). The covers of *Arundinella hirta* and *P. aquilinum* var. *latiusculum*, increased in 2008 at the herbaceous layers of intermediate and low cover plots. The covers of *Q. mongolica*, *Q. serrata*, and *Q. variabilis* at the herbaceous layers in the high cover plots increased by roughly 5% in 2008 compared to 2003.

Runoff and Sediment Yields

Runoff yield increased with increasing rainfall amounts (Fig. 2). Runoff yield was much greater in early years compared to later years. The difference in runoff yield was apparent among the three vegetation cover groups. Runoff yield was greatest in every cover group in 2005 when the most extreme rainfall event (500.5 mm) occurred. At this event, runoff yield was 91.0, 75.7, and 28.0 L m^{-2} at the low, intermediate, and high cover plots, respectively. Runoff yield in the high cover plots was lower for every rainfall event compared with low and intermediate cover plots. Runoff yield per event in the low cover plots was 7.9-fold (range 1.9–23.3) and 4.8-fold (range 1.2–13.7) in the intermediate cover plots, both of which were greater than that of the high cover plots for the 11-year period of data collection.

Sediment yield increased with increasing runoff yield (Fig. 3). Sediment yield per event in the low and intermediate cover plots were 9.1-fold (range 0.9–198.7) and 8.2-fold
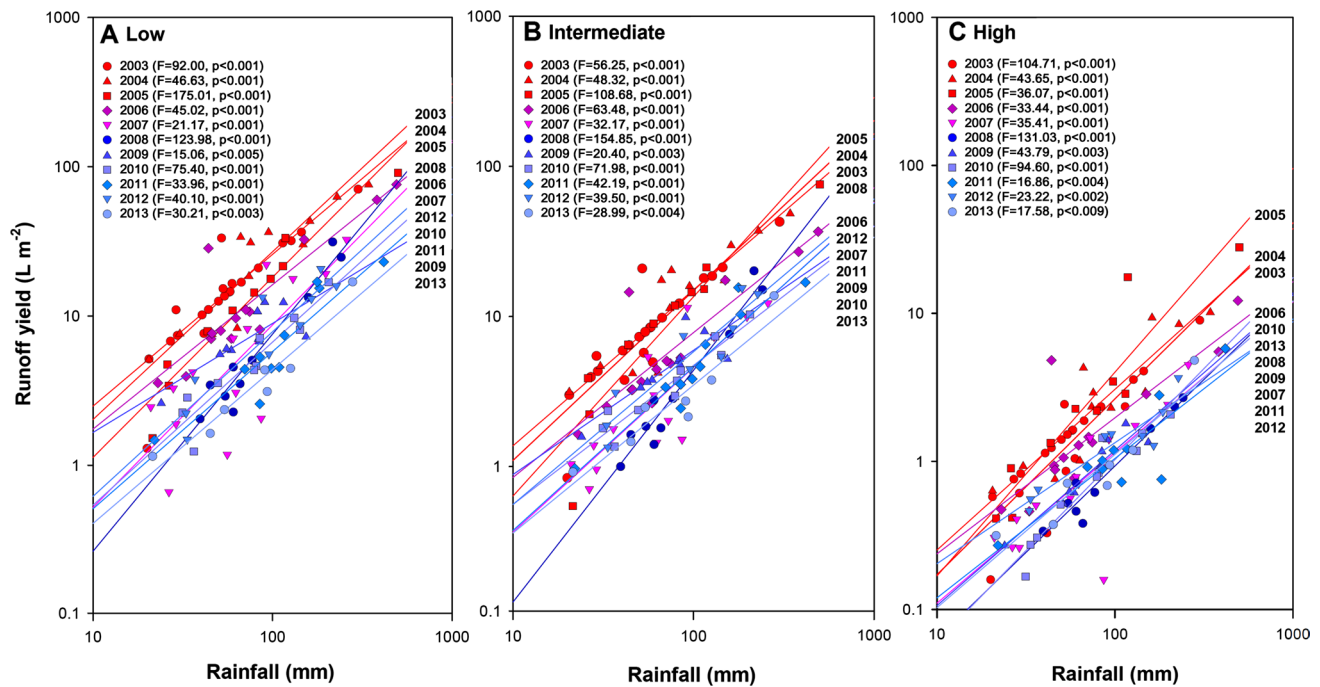
Fig. 2 Relationship between rainfall (mm) and runoff yield (L m^{-2}) at the runoff plots with **a** low, **b** intermediate, and **c** high vegetation cover
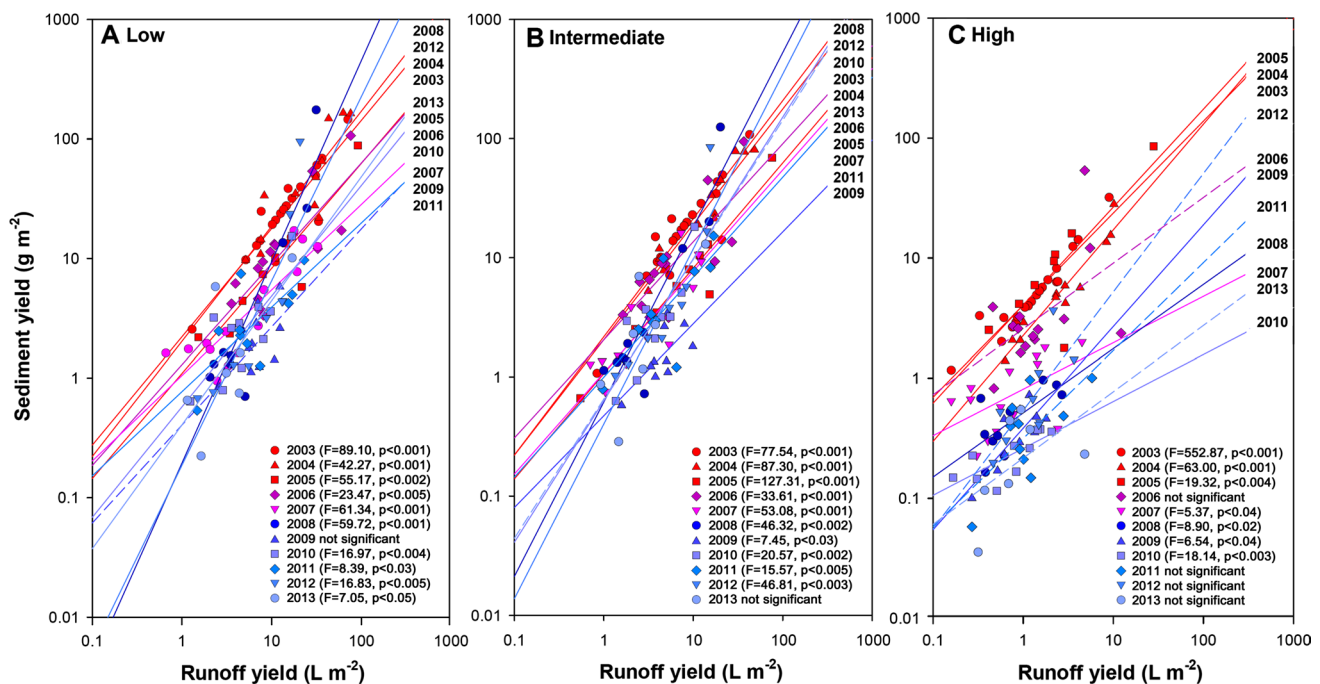

Fig. 3 Relationship between runoff yield (L m^{-2}) and sediment yield (g m^{-2}) at the runoff plots with **a** low, **b** intermediate, and **c** high vegetation cover

(range 0.7–141.9), respectively. Both yields were greater than that of the high cover plots for the 11-year period. Sediment yield was much greater in early years compared to later years, though it fluctuated highly following extreme rainfall

events at the low and intermediate cover plots in particular. For example, in 2008, there was an event with 216.5 mm of rain, and sediment yields of low and intermediate cover plots were 174.7 and 124.8 g m^{-2}, respectively, whereas that of

the high cover plots were only 0.9 g m^{-2}. Additionally, in 2012, there was a rainfall event of 186 mm, and sediment yield was greater for the low (95.2 g m^{-2}) and intermediate (84.8 g m^{-2}) cover plots compared with the high cover plots (3.7 g m^{-2}). At the high cover plots, the sediment yield was the greatest (85.5 g m^{-2}) during the extreme rainfall event in 2005 (500.5 mm). The magnitude of differences between high cover plots and low or intermediate cover plots was greater for sediment yield than runoff yield. Differences in sediment yield between low and intermediate cover plots were not apparent.

Vegetation cover for each year was negatively correlated with: (i) annual runoff yields ($r^2 = 0.43$, $p < 0.001$) and (ii) annual sediment yields ($r^2 = 0.23$, $p < 0.001$) (Fig. 4). Runoff and sediment yield of each runoff plot decreased with increasing years after the fire.

Values for annual runoff yield and sediment yield over time after the fire were fitted with an exponential function. Runoff and sediment yield in the earlier years after the fire decreased greatly over time (Fig. 5a, b). In 2000 (Year 0), runoff yields at the low and intermediate cover plots were estimated to be 825.1 and 455.0 L m^{-2}, 9.1- and 5.0-fold greater than the high cover plots (90.2 L m^{-2}), respectively (Fig. 5a). However, in 2013 (Year 13), those at the low and intermediate cover plots were reduced to 34.0 and 27.2 L m^{-2}, 3.8- and 3.0-fold greater than the high cover plots (9.0 L m^{-2}). Sediment yields at the low and intermediate cover plots in 2000 were estimated to be 2386.9 and 1348.3 g m^{-2}, 6.0- and 3.4-fold greater than the high cover plots (398.4 g m^{-2}) (Fig. 5b). In 2013 (Year 13), those at the low and intermediate cover plots were reduced to 20.3 and 27.3 g m^{-2}, respectively, while there was little sedimentation at the high cover plots (1.9 g m^{-2}).

Cumulative annual runoff yields responding to the rainfall events during the 13-year period were estimated to be 3.77 and 2.21 ton m^{-2} at the low and intermediate cover plots, respectively. These amounts were 7.7- and 4.5-fold greater than the high cover plots (0.49 ton m^{-2}) (Fig. 5c). Cumulative annual sediment yields were estimated to be 2.39, 1.35, and 0.40 kg m^{-2} at the low, intermediate, and high cover plots in 2000 (Year 0), respectively (Fig. 5d). These measures increased remarkably to 5.82, 3.47, and 1.04 kg m^{-2} in 2003 (Year 3), and 7.06, 4.29, and 1.37 kg m^{-2} in 2007 (Year 7). Overall, cumulative annual sediment yields for 13 years were estimated to be 7.53 and 4.72 kg m^{-2}, 5.4- and 3.4-fold greater at the low and intermediate cover plots compared to the high cover plots (1.40 kg m^{-2}) (Fig. 5d). Both cumulative annual runoff and sediment yield rapidly increased during the first few years after the forest fires.

Discussion

Vegetation Recovery

Despite being only three years after the fire, the vegetation has already developed into two strata (i.e. herbaceous and shrub) on three types of plots. The two-layer vegetation is important because it will particularly weaken the impact of raindrops. Vegetation recovery was rapid, in particular at high cover plots, with cover having reached 87% within 3 years after the fires in the present study. At low and intermediate cover plots, however, regeneration was relatively slow, with a maximum of 44% and 69% during the study period. This variability is common in burned forests as post-fire regeneration is strongly influenced by the
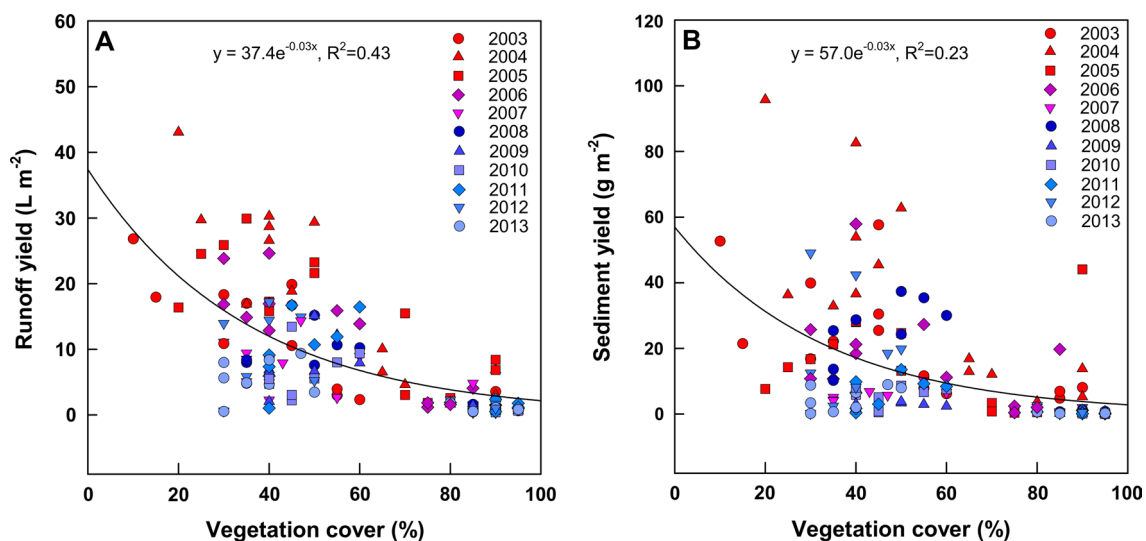

Fig. 4 Relationship between vegetation cover (%) each year and **a** annual runoff yield (L m^{-2}) and **b** sediment yield (g m^{-2})
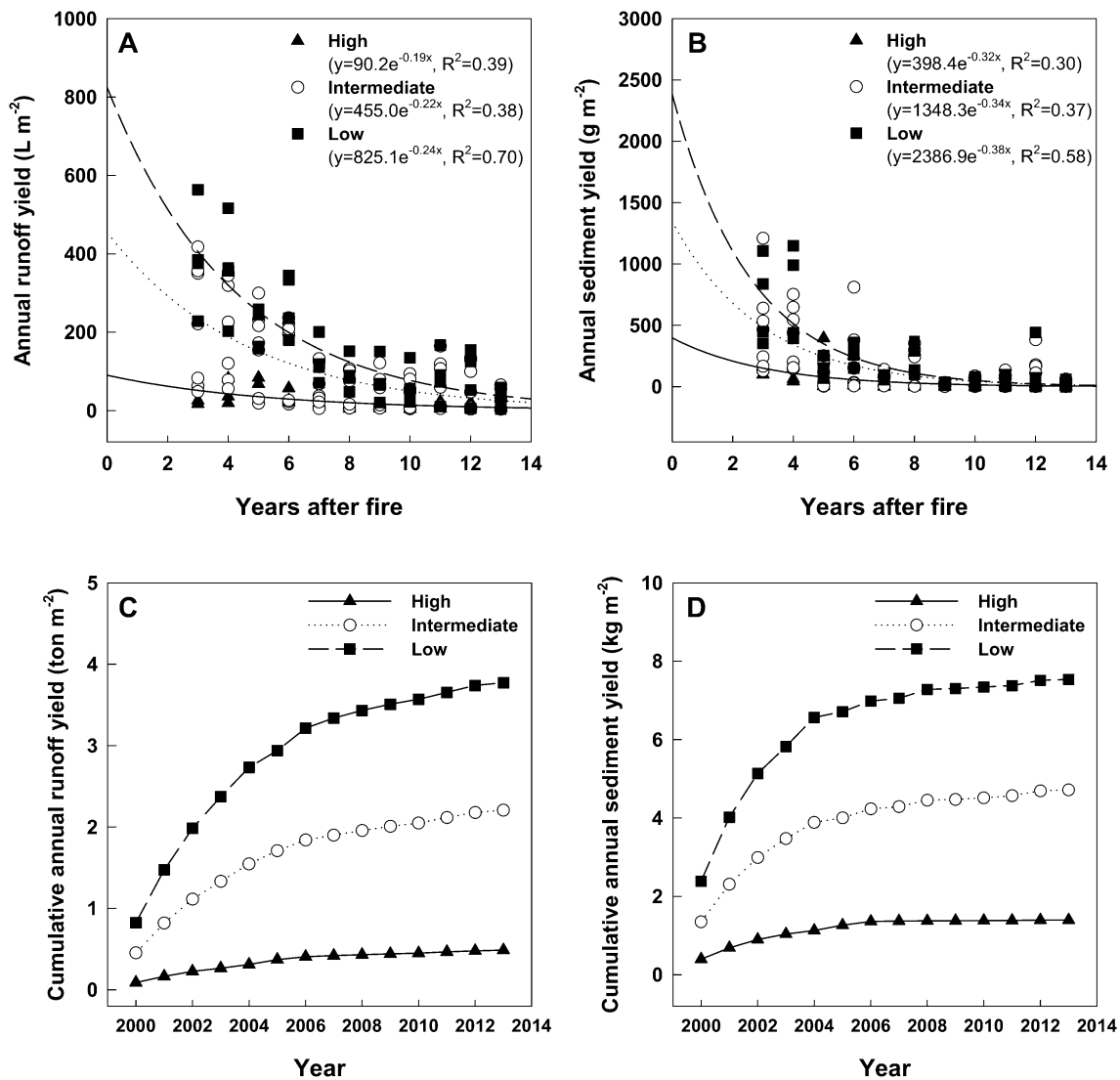
Fig. 5 Relationship between years after fire and **a** annual runoff yield (L m^{-2}) and **b** annual sediment yield (g m^{-2}), **c** cumulative annual runoff yield (ton m^{-2}), and **d** cumulative annual sediment yield (kg m^{-2}) at the runoff plots with low, intermediate, and high vegetation cover

pre-fire composition and quantity of species with sprouting ability (Bellingham et al. 1994; Hodgkinson 1998; Bond and Midgley 2003; Vesk and Westoby 2004). The importance of sprouters in vegetation recovery was clear in the regions affected by the East Coast Fire. Choung and Choung (2019) found that the proportion of sprouter taxa was as high as 81.1%, with non-sprouters comprising only 18.9% of the total in the studied region. As noted by Dietze and Clark (2008), this suggests that vegetative regeneration is the main resilience strategy to disturbances in the region. Specifically, basal sprouting from the root collar of the burned plants (eg. *Q. mongolica* and *L. cyrtobotrya*) and clonal sprouting (eg. *P. aquilinum* var. *latiusculum*) were the most effective strategy for plants.

The results presented here are consistent with studies conducted in Mediterranean regions where fires frequently occur (Gimeno-García et al. 2007; Núñez et al. 2008; Granged et al. 2011). Gimeno-García et al. (2007) observed that following experimental fires in shrubland in Portugal, the site was dominated by resprouting species. After fires in *Pinus sylvestris* forests in Spain, resprouting species, such as *Erica australis* and *Pterospartum tridentatum* belonging to Ericaceae and Leguminosae, respectively, quickly appeared during the first years after the fire (Núñez et al. 2008).

Decrease in Runoff and Sediment Yield with Increasing Vegetation Recovery and Time

Removal of aboveground vegetation due to fire greatly increased overland flow and sediment loss (Cerdá and Doerr 2005; Certini 2005; Durán Zuazo and Rodríguez Pleguezuelo 2008). Although the loss was very large for several years immediately after the forest fire, the magnitude of the response to rainfall events tended to decrease over time remarkably. Since we selected areas with similar topography and soil physicochemical properties, this tendency could be the effect of time and vegetation regeneration. Nonetheless, while soil stabilization was ongoing after the forest fire, sediment was greatly produced corresponding with the extreme rainfall events as shown in 2008 and 2012 (i.e., high rainfall amounts or intensity, or when the rainfall event occurred continuously) at the low and intermediate cover plots. Sheet erosion was observed due to raindrop impact and rill erosion as a result of runoff flow. In some cases, sediment loss at the high cover plots with high root biomass was greater than that of low and intermediate cover plots. This was caused by the abrupt downslope movement of huge soil masses with interconnected root systems as reported at other nearby regions with shallow and erosive soil (Park and Shin 2011; Park et al. 2012; Shin et al. 2013). These characteristics can explain greater variation in sediment yield compared to that in the runoff flow in our study. Complex factors, including not only vegetation cover, rainfall amount, and rainfall intensity but also terrain and topography, can influence soil erosion (Shin et al. 2008).

Vacca et al. (2000) reported that annual soil loss in herbaceous and shrub-dominated vegetation and *Eucalyptus* forests (vegetation cover 25%) was 0.05 and 0.19 kg m^{-2}, respectively, and Romero et al. (1988) suggested large variability with $0.08 \sim 2.55$ kg m^{-2} (cover 35%) of soil loss. Conversely, in vegetation with a cover of 60%, annual soil loss was $0.5 \sim 0.12$ kg m^{-2} (Albadalejo and Stocking 1989), while it was 0.1 kg m^{-2} where the shrub cover was 80% (Bermúdez et al. 1991). Taken together, these results suggest that soil loss is greatly influenced by both cover and type of vegetation.

The Korea Ministry of the Interior and Safety (2019) recommends managing the baseline for sediment loads in forests below 0.12 kg m^{-2} year^{-1}. In our study, sediment yield was estimated to reduce below the baseline at the high cover plots after 2003 (Year 3) and 2007 (Year 7) at both of the intermediate and low cover plots. Overall cumulative sediment loss from low and intermediate cover plots for a 13-year period following the forest fire were estimated 7.53 (5.4-fold) and 4.72 kg m^{-2} (3.4-fold) greater than those of the high cover plots (1.40 kg m^{-2}), respectively. It is likely that the total loss was estimated to be less than the actual loss. We measured runoff and sediment yield from the 3rd year after the fire; however, during the 2nd year after the fire, typhoon Rusa caused serious damage in this area which likely influenced our findings. The role of vegetation to suppress the loss of sediment during the forest regeneration process after the fire was confirmed. A rapid increase of the belowground biomass along with the development of the aboveground would have contributed to the prevention of sediment loss (Lee and Park 2005).

Cerdá and Doerr (2005) suggested that soil surface hydrophobicity is a major contributing factor for decreasing post-fire overland flow. In addition, over the years, herbs, shrubs and trees can increase the depth of the wetting front, and therefore more rainfall could infiltrate the soil, thus reducing the overland flow. Species composition also has a significant impact. Sediment yield per events were fairly high at intermediate cover plots, though not as much was lost as at low cover plots. This is presumed to be the effect of species composition. That is, compared to the low cover plots, the cover of *P. densiflora* and *L. cyrtobotrya* in the shrub layer of intermediate plots was higher, whereas that of *P. aquilinum* var. *latiusculum* was lower. The former two species have needle-leaf and small leaves, respectively, whereas the fern seems to block rainfall due to wide, thick and often multilayered leaves.

Management Implication

The investigation of the natural regeneration of all affected areas (23,794 ha) after the East Coast fire showed that 19% and 47% of total areas belonged to the low and intermediate cover groups, respectively (Choung et al. 2004). With the physical stabilization of the soil and development of vegetation over time, the sediment loss tends to decrease. However, natural recovery of soil is a very slow process during which the soil condition may worsen. Our study found that it took seven years for low and intermediate cover plots, and 3 years for high cover plots for soil loss to decrease below baseline. We estimated that a considerable amount of soil was lost at the low and intermediate cover plots (7.06 and 4.29 kg m^{-2}, respectively) compared to the high cover plots (1.37 kg m^{-2}) over a 7-year period. Cerdà and Lasanta (2005) and Pierson et al. (2008) stated that 3–10 years are typically required to return to pre-fire conditions for runoff and erosion on fire-affected hillslopes. Park et al. (2005) reported that soil was stabilized against erosion within 5 years in fire-affected hillslopes in Korea.

Therefore, for those slow recovery sites, appropriate measures are needed to protect soil against erosion (Shin et al. 2008). We observed from the previous study that wood chip mulching significantly reduced runoff and sediment yield where vegetation was partially recovered (Kim et al. 2008). In addition, straw mulching, hydroseeding, vegetation sack treatment, and log erosion barriers were also effective at

reducing sediment yield on the hillslopes affected by a forest fire (Robichaud et al. 2000; Lee et al. 2014; Lucas-Borja et al. 2019). For the mulching treatment, our results confirm that the bare hillslopes should be mulched to cover roughly 70% of the ground immediately after fire, corresponding to the high cover of this study (Elwell and Stocking 1976; Park et al. 2012; Shin et al. 2013).

Salvage logging and tree planting soon after a forest fire should be reconsidered as these practices significantly increased soil erosion in fire-affected erosive montane forests in a monsoon climate. Shin et al. (2013) investigated sediment yields of hillslopes that were salvage-logged and planted with *P. densiflora* seedlings in 2002, two years after the fire. They found that the maximum sediment yield was 12.5 kg m^{-2} year^{-1}. Significant sediment loss resulted mainly from disturbances in the soil surface caused by management treatment and from the effects of the extreme event of Typhoon Rusa in 2002. Therefore, it is essential that reforestation should be postponed until the levels of vegetation regeneration and soil stabilization are assessed.

Conclusion

Due to the monsoon climate, Korea is very dry in spring and heavy rains are concentrated in summer. Besides, the forest is repeatedly disturbed, resulting in erosive and shallow soil. Under these conditions, soil erosion will be inevitable in the summer following fires in the spring. To explore the long-term dynamics of soil erosion under these regimes, runoff and sediment yield were measured for 11 years (2003–2013) on hillslopes with different vegetation recovery in a pre-fire *P. densiflora* forest. The forest was affected by a stand-replacing fire on the East Coast of South Korea, which occurred in 2000.

Our study confirmed that vegetation is effective in preventing soil loss. Nevertheless, the amount of loss was enormous in the years immediately following the fire. In the fire year, soil losses at the low and intermediate plots were 6.0- and 3.4-fold greater than the high cover plots. It took seven years at low and intermediate cover plots, and three years at high cover plots to reduce to below baseline load. Consequently, it was estimated that the total amount of soil loss for 7 years was 7.06 and 4.29 kg m^{-2} at low and intermediate cover plots, respectively, and 1.37 kg m^{-2} at the high cover plots. Sediment loss fluctuated more than runoff following extreme rainfall events.

We suggest that soil conservation should be a priority when determining policy for the management of the fire-affected hillslopes where soil is erosive and shallow in the monsoon climate. We stress two aspects: (i) if the initial vegetation cover is less than ca. 70%, corresponding to the high cover of this study, appropriate measures, such as wood chip mulching, straw mulching, hydroseeding, vegetation sack treatment, and log erosion barriers, should be introduced to prevent serious soil erosion and to stabilize the soil, (ii) if the initial vegetation cover is greater than ca. 70% and salvage logging and tree planting is necessary, then it should be postponed for at least three years to allow soil to stabilize.

Materials and Methods

Study Site

This study was conducted on hillslopes in Samcheok-si, Gangwon-do, Korea (37°12′43″ N, 129°18′43″E, elevation: ca. 240 m). This site, on the eastern coast of South Korea, was severely burnt by stand-replacing fires in April 2000, an event which damaged 23,794 ha of young forests dominated by 20–40 year old *Pinus densiflora* (Choung et al. 2004). *Pinus densiflora* forests are distributed predominantly in both the east coast regions and across the country (Lee 2010). Most forest fires ignite from *P. densiflora* forests (Lee et al. 2005) which are particularly vulnerable to forest fires (Seo and Choung 2010, 2014). The parental rocks of the site are metamorphic rocks (Geological and Mineral Institute of Korea 1973) and the soil consists of erosive Inceptisol (Rural Development Administration 2019).

We obtained precipitation data from an automatic weather station (8.2 km away) in Wonduk-eup, Samcheok-si (Korea Meteorological Administration 2019). During the study period, the mean annual rainfall was 1339 mm [range: 1801 mm (2006) and 804 mm (2013)] (Fig. 6a). As characteristic of a monsoon climate, roughly 53% and 21% of annual rainfall occurred within the 3rd (July–September) and 2nd quarters (April–June) of the study period, respectively. While there is a trend suggesting decreases in annual rainfall, there is no sign of any trend in rainfall per event or rainfall intensity over the years. The highest maximum rainfall intensity for a 1 h duration was 61 mm h^{-1} in 2008, followed by 53 mm h^{-1} in 2004 (Fig. 6b). Frequencies of rainfall events with > 20 mm h^{-1} of rainfall intensity accounted for 15.7% of total rainfall events.

This study was conducted from 2003 (Year 3 after the fires) to 2013 (Year 13 after the fires). During the plant-growing season in the study period, a total of 127 rainfall events (≥ 20 mm per event) occurred.

Runoff Plots and Field Investigation

In June 2003 (Year 3), 14 runoff plots (10 m long by 3 m wide) were installed. Plot elevations ranged from 247 to 287 m with a similar slope gradient (mean slope of 30°). The contents of sand, clay, and silt were 71.2%, 11.2%, and 17.6%, respectively. Mean soil depth was 34.3 cm. We

Fig. 6 a Annual rainfall (mm) by quarter and **b** maximum hourly rainfall intensity during rain events (mm h^{-1}) of study sites from 2003 to 2013, 3 to 13 years after the east-coast forest fire in 2000
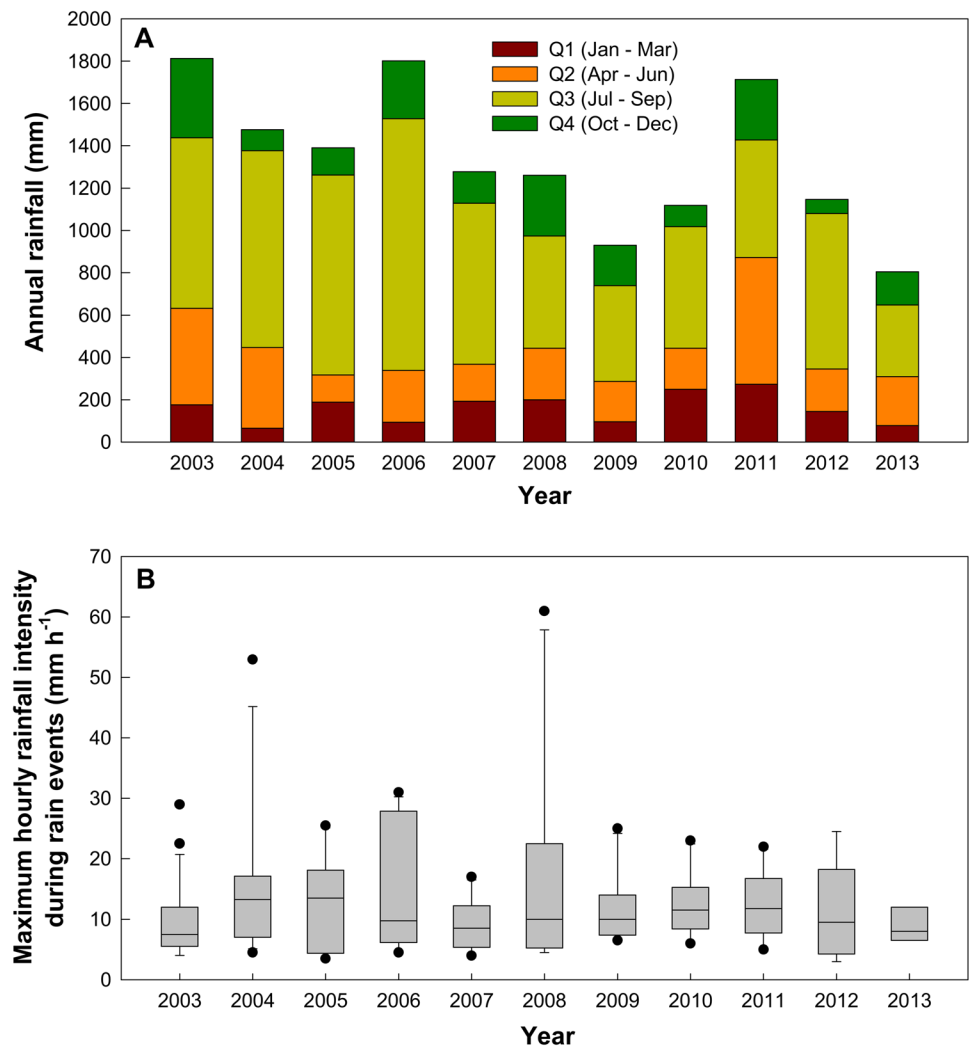
selected areas with: (i) similar burn severity, soil physico-chemical properties, and pre-fire *P. densiflora* stands, but (ii) different vegetation cover.

The runoff plots (10 m long by 3 m wide) were enclosed by steel frames. At the bottom of each plot, a 3 m-wide collecting trough was installed and a 1000 L or 2000 L plastic tank was connected to the trough to collect runoff and sediment produced during each rainfall event. After each major rainfall event (≥ 20 mm), the total volume of runoff and the total weight of sediment were measured in the field, respectively. We transported a 1 L subsample of the sediment to the laboratory where it was dried, weighed, and the weight of the total sediment was estimated. Using linear regression of data for each plot and year, we extrapolated runoff and sediment yield data from tanks that were overflowing due to heavy rainfall.

From 2003 to 2013, the total projected vegetation cover of runoff plots was measured every August. Based on the vegetation cover measured in August 2003, 14 runoff plots were divided into three groups: (i) > 66.7% cover

($n = 3$), hereinafter referred to as "high cover plots," (ii) 33.3% ~ 66.7% cover ($n = 7$), hereinafter referred to as "intermediate cover plots," and (iii) < 33.3% cover ($n = 4$), hereinafter referred to as "low cover plots." Species composition and coverage of shrub (> 0.5 m tall) and herbaceous layers were investigated in 2003 and 2008, respectively. Shrubs and trees in nine runoff plots [high cover plots ($n = 1$), intermediate cover plots ($n = 4$), and low cover plots ($n = 4$)] were cut down by landowners in October 2011.

Statistical Analyses

Linear regression between rainfall and runoff yields, and runoff yields and sediment yields was analyzed using Sigma Plot (version 12.0; SPSS, USA). Plots of runoff yield and sediment yield with increasing years since fire were fit using Sigma Plot. Using the non-linear model, we estimated the runoff yield and sediment yields 0 to 3 years after the fires and calculated cumulative annual runoff and sediment yields in the 11 years since the fires.

Compliance with ethical standards

Conflicts of interest The authors declare no conflicts of interest.

References

Albaladejo MJ, Stocking M (1989) Comparative evaluation of two models in predicting storm soil loss from erosion plots in semi-arid Spain. CATENA 16:227–236

Badía D, Martí C (2008) Fire and rainfall energy effects on soil erosion and runoff generation in semi-arid forested lands. Arid Land Res Manag 22:93–108

Bellingham PJ, Tanner EVJ, Healey JR (1994) Sprouting of trees in Jamaican montane forests, after a hurricane. J Ecol 82:747–758

Bermúdez F, Perez L, Martínez-Fernández J, Díaz A, Sanleandro P (1991) Escorrentías y pérdidas de suelo en Calcisol Pétrico bajo un ambiente mediterráneo semiárido. Cuaternario y Geomorfología 5:77–89

Bond WJ, Midgley JJ (2003) The evolutionary ecology of sprouting in woody plants. Int J Plant Sci 164:S103–S114

Cerdá A, Doerr SH (2005) Influence of vegetation recovery on soil hydrology and erodibility following fire: an 11-year investigation. Int J Wildland Fire 14:423–437

Cerdá A, Lasanta T (2005) Long-term erosional responses after fire in the Central Spanish Pyrenees. 1 Water and sediment yield. CATENA 60:59–80

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

Choung Y (2002) Forest fires and vegetation responses in Korea. In: Lee D, Jin V, Choe JC, Son Y, Yoo S, Lee HY, Hong SK, Ihm BS (eds) Ecology of Korea. Bumwoo Publishing Co, Seoul, pp 119–137

Choung Y, Choung MS (2019) Biodiversity of burned forests is controlled by the sprouting ability of prefire species in *Pinus densiflora* forests. Ecol Eng 126:356–362

Choung Y, Lee BC, Cho JH, Lee KS, Jang IS, Kim SH, Hong SK, Jung HC, Choung HL (2004) Forest responses to the large-scale east coast fires in Korea. Ecol Res 19:43–54

Choung Y, Lee J, Cho J, Noh J (2020) Review on the succession process of *Pinus densiflora* forests in South Korea: progressive and disturbance-driven succession. J Ecol Environ 44:16

Dietze MC, Clark JS (2008) Changing the gap dynamics paradigm: vegetative regeneration control on forest response to disturbance. Ecol Monogr 78:331–347

Durán Zuazo VH, Rodríguez Pleguezuelo CR (2008) Soil-erosion and runoff prevention by plant covers. A review. Agron Sustain Dev 28:65–86

Elwell HA, Stocking MA (1976) Vegetal cover to estimated soil erosion hazard in Rhodesia. Geoderma 15:61–70

Geological and Mineral Institute of Korea (1973) Geological Map of Korea. Hollym Corporation, Seoul

Gimeno-García E, Andreu V, Rubio JL (2007) Influence of vegetation recovery on water erosion at short and medium-term after experimental fires in a Mediterranean shrubland. CATENA 69:150–160

Granged AJP, Zavala LM, Jordán A, Bárcenas-Moreno G (2011) Post-fire evolution of soil properties and vegetation cover in a

Mediterranean heathland after experimental burning: a 3-year study. Geoderma 164:85–94

Hodgkinson KC (1998) Sprouting success of shrubs after fire: height-dependent relationships for different strategies. Oecologia 115:64–72

Kim CG, Shin K, Joo KY, Lee KS, Shin SS, Choung Y (2008) Effects of soil conservation measures in a partially vegetated area after forest fires. Sci Total Environ 399:158–164

Korea Meteorological Administration (2019) Open weather data portal. https://data.kma.go.kr/cmmn/main.do. Accessed 19 June 2019

Korea Ministry of the Interior and Safety (2019) Guidance on consultation practice for disaster impact assessment. Notification No. 2019–5 of the Ministry of the Interior and Safety, Korea

Lee CS, Hong SK (1998) Changes of landscape pattern and vegetation structure in rural area disturbed by fire. Korean J Ecol 21:389–399

Lee CW, Lee CY, Kim JH, Youn HJ, Choi K (2004) Characteristics of soil erosion in forest fire area at Kosung, Kangwondo. Jour Korean For Soc 93:198–204

Lee CW, Seo JI, Youn HJ, Kim K (2014) Effectiveness of rehabilitation treatments on a slowly revegetating hillslope in a recently burned coastal forest, Republic of Korea. Landsc Ecol Eng 10:249–260

Lee DK (2010) Korean forests: Lessons learned from stories of success and failure. Korea Forest Research Institute, Seoul

Lee KS, Park SD (2005) Relationship between the aboveground vegetation structure and fine roots of the topsoil in the burnt forest areas, Korea. Korean J Ecol 28:149–156

Lee SY, Won MS, Han SY (2005) Developing of forest fire occurrence danger index using fuel and topographical characteristics on the condition of ignition point in Korea. Fire Sci Eng 19:75–79

Lucas-Borja ME, González-Romero J, Plaza-Álvarez PA, Sagra J, Gómez ME, Moya D, Cerdà A, de las Heras J (2019) The impact of straw mulching and salvage logging on post-fire runoff and soil erosion generation under Mediterranean climate conditions. Sci Total Environ 654:441–451

Núñez MR, Calvo L, Pando V, Bravo F (2008) Floristic changes induced by fire on *Pinus sylvestris* plantations in northwest of Spain. Inv Agrar Sist Rec F 17:168–177

Park SD, Lee KS, Shin SS (2012) Statistical Soil Erosion Model for burnt mountain areas in Korea – RUSLE approach. J Hydrol Eng 17:292–304

Park SD, Shin SS (2011) Applying evaluation of Soil Erosion Models for Burnt Hillslopes – RUSLE, WEPP and SEMMA. J Kor Soc Civil Eng 31:221–232

Park SD, Shin SS, Lee KS (2005) Sensitivity of runoff and soil erosion in the burnt mountains. J Korea Water Resour Assoc 38:59–71

Pierson FB, Robichaud PR, Moffet CA, Spaeth KE, Hardegree SP, Clark PE, Williams CJ (2008) Fire effects on rangeland hydrology and erosion in a steep sagebrush-dominated landscape. Hydrol Process 22:2916–2929

Robichaud PR, Beyers JL, Neary DG (2000) Evaluating the effectiveness of postfire rehabilitation treatments. General technical report RMRS-GTR-63. United States Department of Agriculture Fores Service, Rocky Mountain Research Station, p 85

Romero R, Guijarro JA, Ramis C, Alonso S (1988) A 30-year (1964–1993) daily rainfall data base for the Spanish Mediterranean regions: first exploratory study. Int J Climatol 18:541–560

Rural Development Administration (2019) Korean Soil Information System. Available at: https://soil.rda.go.kr/. Accessed 19 June 2019

Seo H, Choung Y (2010) Vulnerability of *Pinus densiflora* to forest fire based on ignition characteristics. J Ecol Field Biol 33:343–349

Seo H, Choung Y (2014) Enhanced vulnerability to fire by *Pinus densiflora* forests due to tree morphology and stand structure in Korea. J Plant Biol 57:48–54

Shin SS, Park SD, Cho JW, Lee KS (2008) Effects of vegetation recovery for surface runoff and soil erosion in burned mountains, Yangyang. J Kor Soc Civil Eng 28:393–403

Shin SS, Park SD, Lee KS (2013) Sediment and hydrological response to vegetation recovery following wildfire on hillslopes and the hollow of a small watershed. J Hydrol 499:154–166

Turner MG, Romme WH, Gardner RH, Hargrove WW (1997) Effects of fire size and pattern on early succession in Yellowstone National Park. Ecol Monogr 67:411–433

Vacca A, Loddo S, Ollesch G, Puddu R, Serra G, Tomasi D, Aru A (2000) Measurement of runoff and soil erosion in three areas under different land use in Sardinia (Italy). CATENA 40:69–92

Vesk PA, Westoby MK (2004) Sprouting ability across diverse disturbance and vegetation types worldwide. J Ecol 92:310–320

Wittenberg L, Malkinson D, Barzilai R (2014) The differential response of surface runoff and sediment loss to wildfire events. CATENA 121:241–247