

In situ reclamation of closed coal mine waste in Korea using coal ash

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Abstract Adding coal ash to acid waste decreases its acidity and supplies essential nutrients to plants. The effects of coal ash on acid coal mine waste and acid mine drainage were investigated in the field. Treatments were mixtures of coal mine waste with coal ash (0, 20, and 40%). Addition of coal ash increased the pH of coal mine waste and leachate by 3.11 and 3.03 units, respectively. After stabilization, the concentrations in the leachate decreased from 4.65 to 0.44 mg L⁻¹ for Fe and from 0.31 mg L⁻¹ to “not detected” for Pb. The organic matter content and cation exchange capacity increased significantly (approximately twofold each), for coal mine waste with coal ash. Moreover, plant growth on coal mine waste that had been treated with coal ash was greatly stimulated. The application of coal ash offers an environmentally compatible, cost-effective way to remediate coal mine waste or leachate contaminated with heavy metals.

Keywords Coal ash · Coal mine waste · Heavy metals · Plant growth · Remediation

Introduction

In Korea, from the late 1960s to the late 1980s, a policy to develop coal mining contributed to national industrialization. However, domestic coal mine development has begun

to decline as rapid industrialization has driven a shift to oil and gas-based energy sources [1–3]. There are currently about 2800 domestic metal mines registered; among them are about 500 operating mines, 100 closed mines, and 2200 abandoned mines. Of the 400 registered coal mines, it is reported that about 390 have been abandoned [4]. From this circumstance, closed and/or abandoned metal and coal mines left exposed to the environment without adequate protection might not only undermine the surrounding natural scenery, but could also adversely affect ecosystems due to human disturbance and the proliferation of harmful metals by chemical weathering [5, 6].

Main sources causing detrimental effects on the environment from closed and/or abandoned metal and coal mines are stacks of mine waste and tailings containing sulfide minerals such as pyrite, chalcopyrite, and arsenopyrite. This sulfide-containing mine waste and tailings can generate acid mine drainage (AMD) because when they are exposed to the environment, they lower the pH, thereby releasing H⁺ and reacting with Fe, Al, and SO₄²⁻ via oxidation [2, 5, 7, 8].

Accordingly, the government has carried out various remediation techniques to prevent AMD generation and to restore damaged forestlands near the mines. In the case of AMD treatment, neutralizing materials such as limestone, lime, dolomite, and steel slag have been used to convert the dissolved heavy metals to insoluble form. By adding alkaline amendment, it increases alkalinity of the AMD and generates precipitation of insoluble metal hydroxides combining with CO₃²⁻ and HCO₃⁻ [3, 9, 10].

Forest restoration is another technique to remediate closed and/or abandoned metal and coal mines. In the case of forest restoration, simple covering with clean soil and vegetation has been applied over mine refuse. However, this simple restoration technique can generate problems

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such as poor growth or plant death due to heavy metal toxicity and nutrient deficiency in soil. Therefore, knowledge of the conditions necessary for plant growth including nutrients, drainage, heavy metal content, and plant rooting is in urgent need of improvement for effective forest restoration [11].

Main purpose of this research was to evaluate the stabilization of heavy metals in soil polluted by mine waste with in situ application of coal ash and to prevent release of mine waste with vegetation covering technique.

Materials and methods

Sample collection and analysis

The study areas are located at the Dongwon and Woosung coal mines in Gangwon Province (Sabuk-eup, Jeongseong-gun), where mine refuse has been widely distributed along the mountain slope during past mining activity. This undermines the natural landscape in the region and has adverse effects on ecosystems because of the contaminants leaching out as mine drainage [2]. The experimental material used to stabilize the coal mine waste and mine drainage was a 1:1 (wt wt⁻¹) mixture of fly ash and bottom ash. The fly ash and bottom ash were generated by coal combustion at the Yeongdong Power Plant. The coal mine waste and coal ash for this research were characterized using the Korean standard methods for environmental pollution of soil [12]. Harmful heavy metals, as well as Fe and Al, were also analyzed based on the Korean standard method of environmental pollution for soil [12]. The pH, electrical conductivity (EC), organic matter, and exchangeable cations of the experimental mixture were analyzed based on the methods of soil chemical analysis [13]. An extract from a 1:5 (w v⁻¹) mixture of the experimental materials and distilled water was used to measure the pH and EC. Organic matter content was determined using the Walkley–Black method. The concentration of exchangeable cations was measured in an extract of 1 M ammonium acetate.

On-site field experiment

The buffer curve method was used to determine the optimum mixing ratio between coal mine waste and coal ash. As a result, the amount of coal ash required to correct the pH (6.5) of mine refuse was calculated to be 20% (w w⁻¹) in the laboratory experiment [13]. However, 40% (w w⁻¹) of coal ash was applied in the field experiment due to difficulty in uniform mixing, loss to drainage through large pores, and loss caused by rainfall. Each treatment plot was 28.5 m long, 3.0 m wide, and 1 m deep, and detailed

description of each plot is summarized in Table 1. The lower part of a field lysimeter was set to collect runoff and leachate. Perennial ryegrass (*Lolium perenne*) and Kentucky bluegrass (*Poa pratensis*) seeds were used to introduce vegetation to the area, taking into account their adaptability to the regional climate. Fertilizer was sprayed over the surface of each treatment.

On-site field experiment monitoring

The chemical characteristics of the mine waste, the water quality of the runoff and leachate from the waste, and the growth of introduced vegetation were monitored from September 2009 to August 2012. Samples of mine waste were collected seasonally (March, June, September, December) from the topsoil (0–15 cm) with hand auger, and five subsamples in each plot were combined to make one composite sample in a plastic bag. Collected samples were transported into a laboratory, air-dried, and ground to pass through a 2-mm sieve for chemical analysis. For heavy metal analysis, air-dried samples were sieved with 0.15-mm sieve [12, 14]. Runoff and leachate were collected in a container separately, and each sample was collected 1 day after rainfall event. One liter of runoff and leachate samples was contained in a plastic sample bag and transported into a laboratory with an ice cooler. The water samples were stored in a refrigerator after hydrogen nitrate (HNO₃) was added to make the pH < 2 for cation analysis [2]. The pH and EC of the runoff and leachate were measured on-site. The heavy metal contents of the water samples and mine waste were determined using inductively coupled plasma (ICP, Thermo, iCAP 6000 series). The cation contents were analyzed using an atomic absorption spectroscopy (AAS, PerkinElmer, PinAAcle 900F). In addition, rainfall amount and air temperature are summarized in Fig. 1.

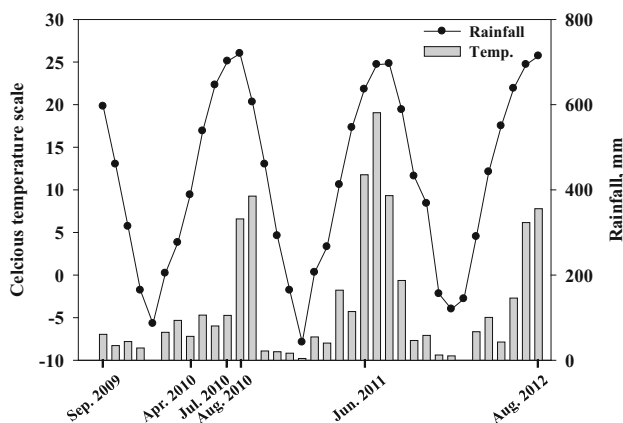
Results and discussion

Chemical properties and heavy metal concentration of coal mine waste and coal ash

Detailed chemical properties and heavy metal concentration of both coal mine waste and coal ash are summarized in Table 2. The coal mine refuse was characterized in terms of pH (3.1), Al (205 mg kg⁻¹), and Fe (498 mg kg⁻¹). Considering the highly acidic condition, the leachate (which included Al and Fe) was expected to result in “yellow boy” and aluminum whitening symptoms in rivers and streams [2, 3, 8]. The chemical properties of pH, organic matter (1.6 g kg⁻¹), and available phosphate (4.2 mg kg⁻¹) were likely to cause nutrient shortages and

Table 1 Description of treatment for field experiment

Treatment	Description
W	Mine waste only
WC20M	Mine waste and coal ash 20% (w w ⁻¹): completely mixed
WC40M	Mine waste and coal ash 40% (w w ⁻¹): completely mixed
WC40C	Mine waste and coal ash 40% (w w ⁻¹): covered (bottom: mine waste, top: coal ash)
WC40MD	Mine waste and coal ash 40% (w w ⁻¹): covered (bottom: completely mixed mine waste and coal ash, top: fresh soil 10% (w w ⁻¹))
WC40D10M	Mine waste and coal ash 40% (w w ⁻¹) + fresh soil 10% (w w ⁻¹): completely mixed

**Fig. 1** Average temperature and rainfall amount during experimental period**Table 2** Chemical properties and heavy metal concentration of coal mine waste and coal ash

Chemical properties	Coal mine waste	Coal ash
pH (1:5)	3.1 ± 0.05	10.6 ± 0.36
Organic matter (g kg ⁻¹)	1.6 ± 0.06	5.9 ± 0.062
Avail P ₂ O ₅ (mg kg ⁻¹)	4.2 ± 0.78	241.3 ± 16.
CEC (cmol ₍₊₎ kg ⁻¹)	12.86 ± 0.06	21.9 ± 0.13
Heavy metal concentration (aqua regia, mg kg ⁻¹)		
As	217 ± 15.8	13.3 ± 0.39
Cd	1.66 ± 0.12	0.21 ± 0.01
Cu	4.83 ± 0.43	24.8 ± 0.48
Pb	83.8 ± 8.18	18.3 ± 0.19
Zn	24.3 ± 3.05	140 ± 1.09
Ni	12.5 ± 0.57	20.9 ± 0.06

thus inhibit plant growth, as was reported in previous studies [15]. Vegetation in the study areas was small or dead where the coal mine wastes were distributed. The coal ash used to improve the characteristics of the mine wastes and to stabilize heavy metals was analyzed and found to have a pH of 10.6, EC of 1.690 dS cm⁻¹, exchangeable cations such as Ca²⁺ (85.1 mg kg⁻¹), Mg²⁺ (14.3 mg kg⁻¹), K⁺ (8.6 mg kg⁻¹), available phosphate

(P₂O₅, 241.3 mg kg⁻¹), and organic matter (5.9 g kg⁻¹). The results showing a higher pH, a large amount of exchangeable ion content, and the organic matter content were considered able to stabilize heavy metals by a variety of mechanisms [2, 3, 16–18]. The large amounts of available phosphate, exchangeable cations, and organic matter were likely to improve the plant growth conditions, to stabilize heavy metals in the coal mine waste, and thus to serve as effective plant media [2, 8, 19].

Changes in chemical properties of mine waste after remediation

When a certain amount of rainfall occurred after the on-site treatments were set up, field samples were collected. In this study, eight samplings were evaluated using the results of data collected over about 4 years. The physicochemical properties of the mine waste collected from fields to which coal ash treatment had been applied (pH, EC, organic matter content, cation exchange capacity) were shown to change (Table 3). The pH of the control (W) remained steady at 3.11 ± 0.2. The pH increased in the order W < WC20M < WC40M < WC40C < WC40MD < WC40D10M for the duration of the monitoring period. For the coal ash treatments applied, the pH was higher, showing improvements of 40% (5.22 ± 0.6–6.96 ± 0.2) or 20% (4.46 ± 0.7), depending on the amount blended. The pH calibration capabilities were higher in WC40D10M (6.53 ± 0.2–6.96 ± 0.2) than in WC40M (5.22 ± 0.6–5.97 ± 0.6) [2, 20]. We suggest that the high neutralizing capacity is due to having a large amount of Ca and Mg oxides in the coal ash [21–23]. The EC was found to maintain a certain higher level early in the study, but gradually stabilized in all treatments. It was expected that a large amount of the ions contained in the mine wastes and coal ash would be released initially, but that the amount of ions released would then gradually stabilize [3, 15].

Organic matter that could be used as a fertility index of the control was maintained at an average of 1.51 ± 0.04 g kg⁻¹ level for the duration of the

Table 3 Change in chemical properties of mine waste with coal ash treatments

pH (1:5)	W	WC20M	WC40M	WC40C	WC40MD	WC40D10M
September 2009	3.09 ± 0.05d	5.30 ± 0.11c	5.94 ± 0.16b	6.76 ± 0.06a	6.64 ± 0.27a	6.91 ± 0.15a
November 2009	2.81 ± 0.14d	5.25 ± 0.23c	5.86 ± 0.17b	6.53 ± 0.17a	6.77 ± 0.33a	6.89 ± 0.19a
April 2010	3.30 ± 0.13e	5.27 ± 0.30d	5.82 ± 0.08c	6.56 ± 0.13b	6.85 ± 0.09a	7.11 ± 0.12a
August 2010	3.01 ± 0.12f	3.80 ± 0.16e	5.03 ± 0.15d	5.54 ± 0.25c	6.29 ± 0.19b	6.73 ± 0.13a
June 2011	3.16 ± 0.09e	4.51 ± 0.09d	4.76 ± 0.07d	5.78 ± 0.24c	6.37 ± 0.18b	6.86 ± 0.18a
November 2011	3.22 ± 0.04a	3.58 ± 0.16b	5.17 ± 0.15c	5.68 ± 0.06d	6.48 ± 0.35e	6.89 ± 0.05f
April 2012	2.89 ± 0.13a	4.05 ± 0.17b	4.91 ± 0.11c	5.33 ± 0.16d	6.43 ± 0.14e	7.33 ± 0.14f
August 2012	3.38 ± 0.15a	3.89 ± 0.18b	4.28 ± 0.09c	5.58 ± 0.08d	6.38 ± 0.04e	6.93 ± 0.05f
OM (g kg ⁻¹)	W	WC20M	WC40M	WC40C	WC40MD	WC40D10M
September 2009	1.61 ± 0.15e	2.84 ± 0.07d	3.61 ± 0.03b	5.19 ± 0.15a	2.98 ± 0.10d	3.26 ± 0.06c
November 2009	1.55 ± 0.09e	2.67 ± 0.16d	3.58 ± 0.16b	4.89 ± 0.14a	3.01 ± 0.06c	3.21 ± 0.09c
April 2010	1.49 ± 0.06d	2.91 ± 0.09c	3.38 ± 0.04a	3.16 ± 0.03b	3.16 ± 0.03c	2.86 ± 0.12c
August 2010	1.51 ± 0.03d	2.83 ± 0.17c	3.34 ± 0.18a	3.21 ± 0.14ab	2.79 ± 0.10c	3.02 ± 0.19bc
June 2011	1.43 ± 0.04e	2.56 ± 0.06d	3.31 ± 0.04a	2.67 ± 0.14cd	2.81 ± 0.12bc	2.86 ± 0.04b
November 2011	1.47 ± 0.04c	2.37 ± 0.21b	2.89 ± 0.13a	2.49 ± 0.07b	2.83 ± 0.19a	2.93 ± 0.19a
April 2012	1.52 ± 0.05d	2.42 ± 0.12c	2.73 ± 0.10ab	2.53 ± 0.23bc	2.76 ± 0.07a	2.87 ± 0.07a
August 2012	1.54 ± 0.03e	2.51 ± 0.11cd	2.68 ± 0.08bc	2.47 ± 0.11d	2.92 ± 0.03a	2.83 ± 0.16ab
CEC (cmol ₊ kg ⁻¹)	W	WC20M	WC40M	WC40C	WC40MD	WC40D10M
September 2009	13.4 ± 0.22f	21.3 ± 0.24c	23.9 ± 0.23b	25.2 ± 0.17a	16.1 ± 0.25e	17.0 ± 0.14d
November 2009	13.3 ± 0.12d	21.0 ± 0.21b	25.4 ± 0.34a	25.3 ± 0.49a	16.3 ± 0.32c	16.9 ± 0.17c
April 2010	12.9 ± 0.06f	22.1 ± 0.32b	25.1 ± 0.15a	20.9 ± 0.30c	15.9 ± 0.36e	17.1 ± 0.03d
August 2010	13.1 ± 0.25f	21.7 ± 0.72b	24.6 ± 0.49a	20.9 ± 0.06c	15.9 ± 0.11e	16.7 ± 0.45d
June 2011	13.2 ± 0.39f	21.8 ± 0.59b	24.9 ± 0.15a	21.1 ± 0.39c	16.0 ± 0.13e	16.9 ± 0.28d
November 2011	12.9 ± 0.24e	20.7 ± 0.16b	24.8 ± 0.29a	20.7 ± 0.13b	15.8 ± 0.51d	16.8 ± 0.17c
April 2012	13.0 ± 0.06e	20.4 ± 0.24b	24.3 ± 0.13a	20.6 ± 0.11b	16.1 ± 0.27d	17.3 ± 0.30c
August 2012	13.0 ± 0.16e	20.7 ± 0.14b	24.3 ± 0.29a	20.5 ± 0.28b	15.8 ± 0.40d	16.9 ± 0.32c

Chemical properties are compared with different treatments and same letters represent that value is not significantly different ($p < 0.05$). Soil chemical properties did not differ significantly ($p < 0.05$) with different treatments

monitoring period. However, the organic matter increased in the order of WC20M ($2.64 \pm 0.21 \text{ g kg}^{-1}$) > WC40MD ($2.87 \pm 0.09 \text{ g kg}^{-1}$) > WC40D10M ($2.99 \pm 0.16 \text{ g kg}^{-1}$) > WC40M ($3.19 \pm 0.37 \text{ g kg}^{-1}$) > WC40C ($3.33 \pm 1.09 \text{ g kg}^{-1}$) in the coal ash treatments. This is expected to reflect the specific characteristics of each level of organic matter content (15%) of the coal ash. In addition, the organic matter content in the WC40C treatments was high ($5.04 \pm 0.21 \text{ g kg}^{-1}$) at the start of the experiment (2009), then gradually decreased, and stabilized at $2.76 \pm 0.34 \text{ g kg}^{-1}$. This appeared to be caused by the coal ash layer over the mine waste layer being lost to rainfall or wind, and because the organic matter in the WC40C material was also lost through macropores at the bottom of the mine waste layer. Our result agrees with the previous studies that the chemical (e.g., organic content of

the mine waste) and physical properties (e.g., bulk density, water-holding capacity) could be improved in accordance with the coal ash application [21–23] (Table 3).

Changes in runoff and leachate characteristics

Figures 2 and 3 show the results of monitoring the characteristic changes in the runoff and leachate collected after the field treatments. In the case of the initial runoff (September 2009), because rainfall did not occur, the sample was not collected in a collecting container. The pH of the runoff in the control showed that the quality of the water draining continued to be 3.56–4.12 (average: 3.92), but in the coal ash treatments, the final pH was increased by the range of 0.74 (for WC40C) to 1.79 units (for WC40D10M). Similarly, leachate was discharged

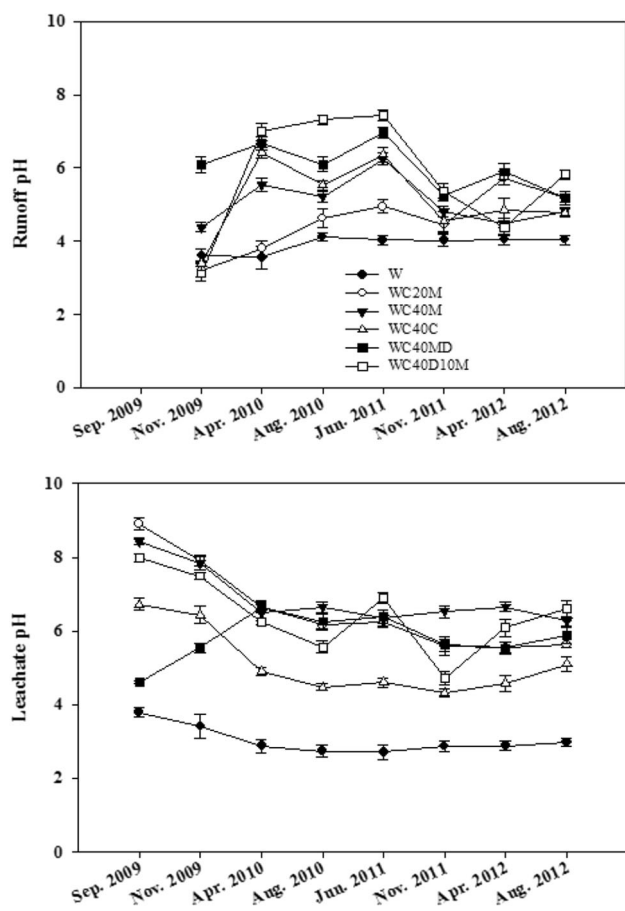


Fig. 2 Change in pH of runoff and leachate as affected by coal ash treatment

continuously and was strongly acidic (pH 2.71–3.79) for W. After the coal ash treatments, the final pH was also increased, by the range of 2.12 (for WC40C) to 3.62 units (for WC40D10M). In previous studies [24, 25], it was reported that coal ash was able to moderate pH. The control at low pH was caused continuously by the dissolution of precipitates of sulfuric acid ions contained in the mine waste and/or by the influence of iron-oxidizing bacteria on the oxidation of sulfide [7, 2]. The pH increased after the coal ash treatments because cations in the coal ash were activated by water [3, 23, 24, 26]. It was determined that this result was the same as reported earlier, i.e., that at the time the pH increased, the amount of eluted Ca and Mg increased [2]. These results suggest the utilization of a high-pH agent for neutralization, such as soil or mine waste with an acidic condition.

The EC of runoff from the coal ash treatments (0.19–0.25 dS m^{-1}) appeared to be high (0.03–0.12 dS m^{-1}), compared to the control. One year after the monitoring started (August 2010), in all treatments, the EC had stabilized to the range of 0.09–0.12 dS m^{-1} . These results show that the ions from the coal ash are expected to leach

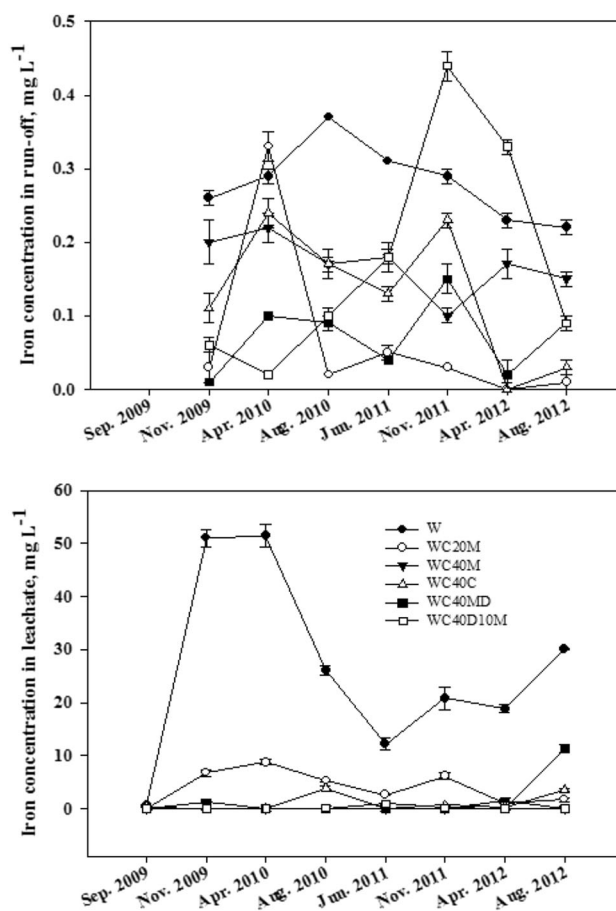
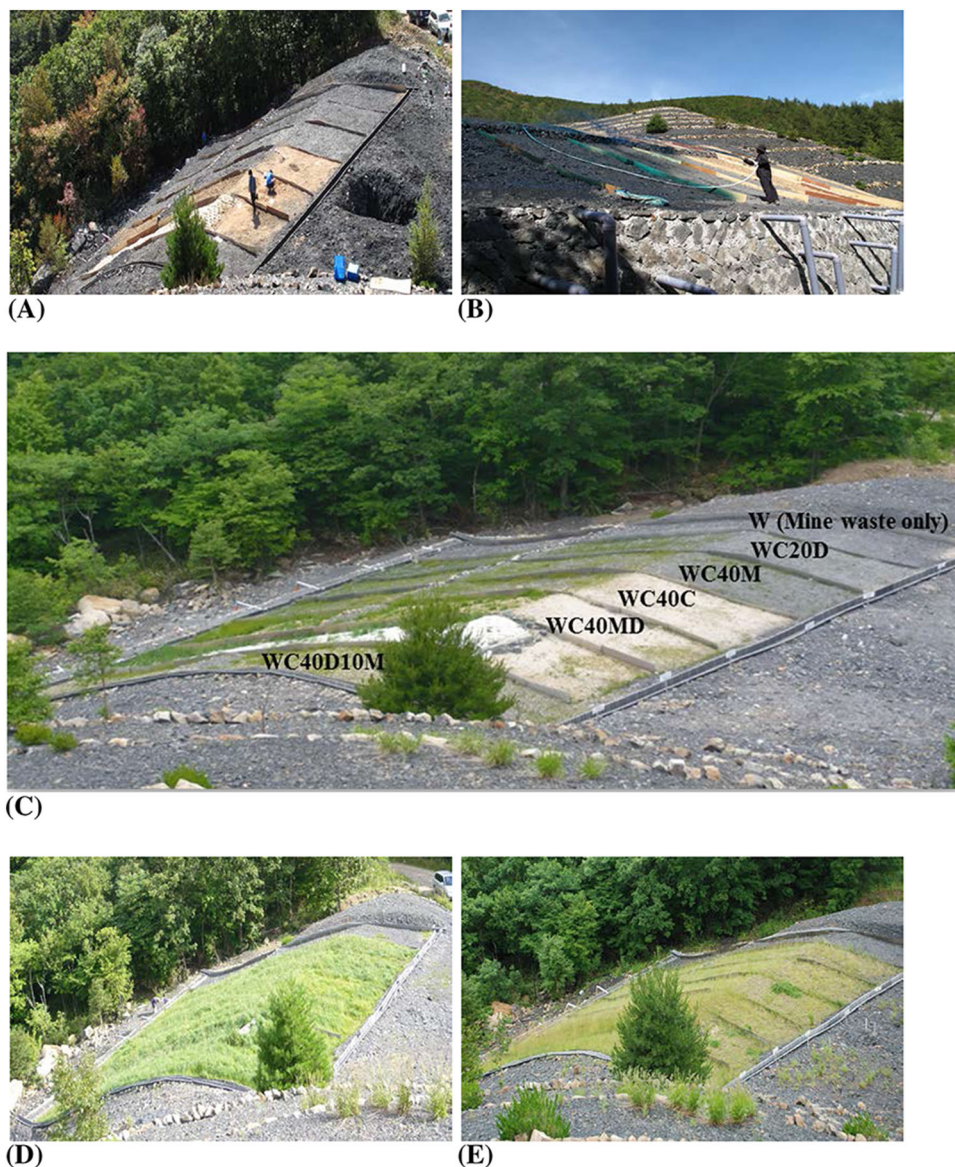


Fig. 3 Change in iron concentration in runoff and leachate as affected by coal ash treatment

under the condition of acidic pH [3, 25]. For the leachate, the average EC level was 0.05 dS m^{-1} for W, WC40MD, and WC40D10M, for which control and cover materials were used. However, when coal ash was used alone, the EC was 0.61 dS m^{-1} (for WC40C), 1.49 dS m^{-1} (for WC20M), and 1.51 dS m^{-1} (for WC40M). These measures were 10 \times higher than for runoff from the same treatments. It is expected that a large number of ions could be leached from coal ash treated with mine waste because of a longer response time [27]. In addition, the coal ash treatments (WC40C) stabilized at 0.3 dS m^{-1} in the third year of monitoring, which was determined to be an effect of the loss of coal ash.

Heavy metal(loid) analysis of the runoff and leachate was carried out for As, Cd, Pb, and Fe. However, only the Fe content is shown in the results (Fig. 3). The runoff results in the treatments were As: ND (not detected)–0.08 mg L^{-1} , Cd: ND, Pb: ND–0.05 mg L^{-1} , and Fe: ND–0.37 mg L^{-1} . It was shown that in the coal ash treatments, many heavy metal(loid)s were not detected or were below the detection limit of the analysis, and the highest concentration of all heavy metal(loid)s was measured in the

Fig. 4 Picture of plant growth in field lysimeter as affected by coal ash treatment, (A) field lysimeter setup (September 2009), (B) seed spray in lysimeter, (C) July 2010 (after 11 months), (D) August 2010 (after 12 months), (E) August 2012 (after 36 months)



control plot (As: 0.08, Pb: 0.05, and Fe: 0.37 mg L⁻¹). Compared to the water quality standards for surface water quality ([28]; Cd 0.01: mg L⁻¹ or less; Cu: no reference; Pb: 0.1 mg L⁻¹ or less; and As 0.05: mg L⁻¹ or less), all treatments, including the control, were found to be safe in terms of Cd and Pb, but As exceeded the reference value in the control. For the leachate, the results were As: ND–0.19 (for W) mg L⁻¹, Cd: ND, Pb ND–0.3 mg L⁻¹, and Fe: ND–51.4 mg L⁻¹. Heavy metal(loid) concentration of leachate was generally 2–15× greater than in the runoff. The average highest concentration of metal(loid)s was also observed in the control plot (As: 0.19, Pb: 0.31, and Fe: 51.4 mg L⁻¹). Because Pb and As exceeded the surface water quality standards, they will likely require application of a stabilization process to reduce risk to the environment. However, in accordance with the details of the coal ash

treatment, the metal ions for the leachate also decreased compared to the control. The reduction of heavy metal(-loid) ions after the coal ash treatments was determined to be consistent with the result that, in the case of As, materials containing a large amount of Ca compounds reacted with As to generate and stabilize water-insoluble compounds (e.g., Ca–As–O compounds) [29, 30]. Moreover, compounds such as CaCO₃ and CaO were precipitated in water-insoluble forms of PbCO₃ by reaction with metal cations [31]. In addition, Yeheyis et al. [25, 32] reported that Si and Al contained in the coal ash were stabilized by the pozzolanic reaction to form calcium–silicate–hydrate (CaO·SiO₂·H₂O) and calcium–alumina–hydrate (CaO·Al₂O₃·H₂O).

For runoff and leachate, the greatest concentration of Fe was detected periodically in the control (runoff average:

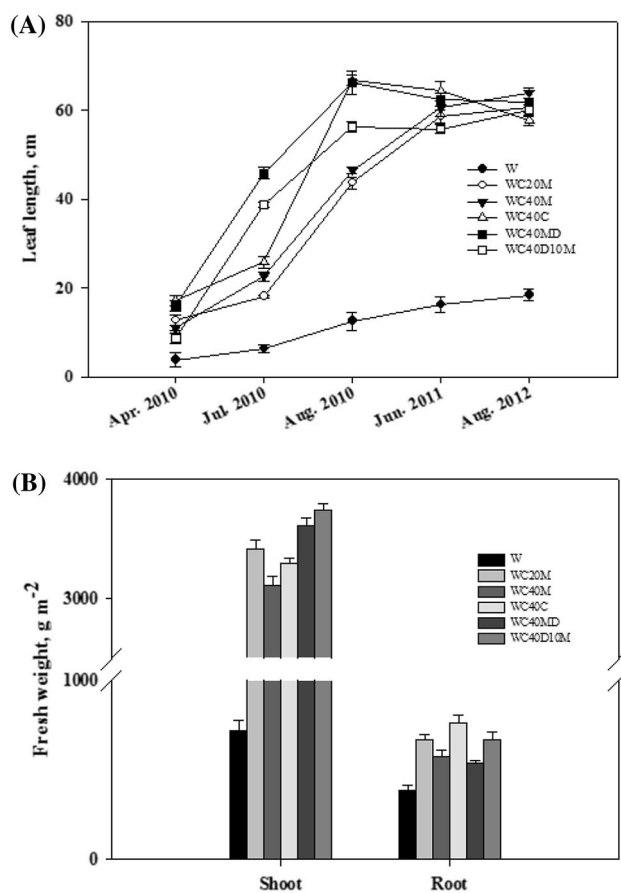


Fig. 5 Change in (A) leaf length and (B) fresh weight in field lysimeter as affected by coal ash treatment

$0.28 \pm 0.05 \text{ mg L}^{-1}$; leachate average: $30.09 \pm 15.52 \text{ mg L}^{-1}$, except for the initial concentration). The Fe content in the leachate was about $100\times$ higher than that in the runoff, which indicated that water-soluble Fe was leached via pores [27]. The Fe content of the runoff during the monitoring period was highest in the control (average $0.28 \pm 0.05 \text{ mg L}^{-1}$) and appeared in the order: WC40D10M ($0.17 \pm 0.16 \text{ mg L}^{-1}$) \approx WC40M ($0.17 \pm 0.04 \text{ mg L}^{-1}$) > WC40C ($0.15 \pm 0.08 \text{ mg L}^{-1}$) > WC20M ($0.08 \pm 0.12 \text{ mg L}^{-1}$) > WC40MD ($0.07 \pm 0.05 \text{ mg L}^{-1}$) for the coal ash treatments. However, the Fe content of the runoff was not significant in the control and the coal ash treatments. The Fe content of the leachate was $51.3 \pm 0.25 \text{ mg L}^{-1}$ in the control after 1 year of treatment. The Fe content gradually stabilized to $21.6 \pm 6.87 \text{ mg L}^{-1}$. It finally averaged $30.1 \pm 15.5 \text{ mg L}^{-1}$, which was far beyond the standards of the US EPA water quality standards for natural water (1 mg L^{-1}), the national drinking water standard (0.3 mg L^{-1}), and the water regulatory standards (clean area 2 mg L^{-1} , “A,” “B,” “C” area 10 mg L^{-1}). Thus, it was

determined that there was a significant level of adverse effects [24, 28]. However, the Fe content of the leachate from the coal ash treatments was reduced in the order: WC20M (to 4.65 mg L^{-1}) > WC40MD (to 1.91 mg L^{-1}) > WC40C (to 1.23 mg L^{-1}) > WC40M (to 0.44 mg L^{-1}) > WC40D10M (0.23 mg L^{-1}). For the coal ash treatments, except for WC20M, the Fe content satisfied the South Korean national water quality regulatory standards; furthermore, WC40M and WC40D10M treatments appeared to have sufficiently purified the water so that it was close to the national drinking water quality standards. It was determined that this improvement was due to water-soluble ferrous ions (Fe^{2+} , Fe^{3+}) that formed insoluble precipitate at pH 5.0 or higher, which was in turn due to the coal ash used in the study to create an alkaline condition that caused water-insoluble precipitates such as $\text{Fe}(\text{OH})_2$, FeCO_3 , and $\text{Fe}(\text{OH})_3$ to form from the reaction of Ca in the coal ash [24, 25, 33].

Restoring forests

Growth monitoring was not conducted the first year (2009) after the seeds were sown. The plant heights were periodically measured in the second year. The fresh weight of vegetation collected from an area of 1 m^2 was evaluated in November (Fig. 4). Field measurements of the vegetation were taken (plant height and fresh weight) to evaluate the treatment results (Fig. 5). According to visual inspection, the degree of land cover was about 10% for the control and complete (100%) for the areas treated with coal ash. The plant height of the vegetation appeared to indicate continuous growth in all treatments; however, when evaluating growth at the final plant height, the plant heights showed that 70% of the growth in the control (18.4 cm) was poor compared to that in the areas treated with coal ash. In addition, production based on the fresh weight was found to be only 19–23% in the control area (719 g m^{-2}), compared to that in the areas treated with coal ash. This was determined to be due to the improvement in the chemical characteristics, such as the reduction of toxic heavy metal ions due to pH improvement (less acidic), the increased supply of nutrient ions required for the synthesis of chlorophyll in the plants, better nitrogen fixation due to increase in the supply of organic material, and increase in the available phosphate due to the application of coal ash in the prior study [34–36].

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