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# Influence of Spoil Type on Chemistry and Hydrology of Interflow on a Surface Coal Mine in the Eastern US Coalfield

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**Abstract** Surface mining for coal is responsible for widespread degradation of water resources and aquatic ecosystems in the Appalachian Region, USA. Because native topsoils are typically not retained on Appalachian mined sites, mine soils are usually composed of crushed overburden. This overburden tends to contribute high salinity loads to downstream aquatic systems. Also, loss of transpiration from forests and reduced infiltration associated with conventional reclamation procedures lead to altered water budgeting and stream morphology. To investigate the influence of the geologic composition of this overburden on water quality and tree growth, a series of experimental plots were constructed on a reclaimed surface mine site in eastern Kentucky, USA, in 2005. Treatments included unweathered GRAY sandstone, weathered BROWN sandstone, and MIXED sandstones and shale spoils. Plots were composed of end-dumped, uncompacted spoils and were designed to drain interflow through data acquisition stations for sampling purposes. Most water chemical parameters had stabilized across all treatments by 9 years after spoil placement. Discharge volume was not different among treatment types through the first 3 years after placement. However, 9 years after placement, seasonal variation in discharge on BROWN is more extreme than that on MIXED or GRAY. In addition, planted tree growth on BROWN has drastically outpaced growth on GRAY or MIXED, suggesting that evapotranspiration may be influencing seasonal variation in water discharged from BROWN. These results suggest that placement of brown weathered spoils when soil substitutes are required may lessen hydrologic impacts via improved tree growth and water utilization on surface-mined sites in Appalachia.

**Keywords** Appalachia · Surface mine reforestation · Mine soils · Water quality · Electrical conductivity

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## 1 Introduction

Surface mining for coal has been tied to a broad spectrum of environmental impairments, both in the Appalachian region of the USA and globally. This controversial mining process removes natural forest soil and vegetative cover and excavates rock overlying target coal seams (overburden). Excess rock is often permanently placed in “valley fills,” burying headwater streams. After coal extraction, overburden is replaced on site and recontoured to a desired topography. Reclamation typically results in heavily compacted mine soils that are often colonized by competitive nonnative vegetative species. Both spoil compaction and competitive groundcovers inhibit the establishment of forests similar to those that occupied the area prior to mining (Zipper et al. 2011).

In addition to extensive impacts on terrestrial ecology, surface mining leads to long-term and widespread water quality degradation, especially with respect to altered hydrology and impaired water chemistry (Bernhardt and Palmer 2011). Altered streamflow is linked both to reduced evapotranspiration (Hornbeck et al. 1970) and reduced infiltration because of soil compaction (Ritter and Gardner 1993). Ferrari et al. (2009) modeled flood response in surface-mined land in Virginia, USA, and observed that flood magnitude increased with increased surface-mined area. In contrast, forestry operations that remove forest cover were found to increase overall water yield, but not increase flood volume (Kochenderfer et al. 2007). Thus, the heavy compaction associated with typical surface mine reclamation produced a flashier hydrograph. Another study in Maryland, USA, (Negley and Eshleman 2006) found that storm event runoff from a watershed influenced by surface mining was significantly higher than runoff from an undisturbed forested watershed. Total annual runoff from the watersheds was similar, however, due to higher baseflow from the forested watershed. The authors attributed this difference in storm response to heavy compaction on the surface-mined watershed, which reduced infiltration rates to <1 cm/hour, compared to 30 cm/hour in the reference. Ritter and Gardner (1993) observed that increased severity of storm response can lead to severe stream morphological consequences, particularly channelization.

The impacts of surface mining on water chemistry are well documented. Much of the water quality impacts observed downstream of surface mining operations are tied to the excavation of large volumes of unweathered overburden. Minerals and salts that were previously bound in impermeable, unweathered rock formations are rapidly exposed to weathering and can be easily dissolved in rainwater. Much research has been conducted investigating the impacts and treatment of acid mine drainage (AMD). This phenomenon occurs when reduced sulfur compounds (e.g., pyritic materials) previously isolated in coal and overburden are exposed to water, air, and specialized soil microbial communities, which oxidize the sulfides to sulfuric acids. This acidic water, in addition to its inherent toxicity, tends to carry elevated loads of metals such as Fe and Al, which contribute to additional toxicity (Soucek et al. 2000; Schmidt et al. 2002; Kennedy et al. 2003). With improved understanding of the processes contributing to AMD, appropriate isolation of pyritic materials during excavation and reclamation can effectively prevent subsequent oxidation and acidification. Because of this, most mine sites currently in operation do not have issues with acidic mine drainage; instead, high carbonate levels in deep, unweathered geological strata increase pH of draining water, thus providing alkaline pH stress (Lindberg et al. 2011).

A clear biological indicator of water toxicity associated with surface mining is macroinvertebrate community impairment. Pond et al. (2008) found that sensitive macroinvertebrate taxa (Ephemeroptera, Plecoptera, and Trichoptera) were largely extirpated from streams draining surface mines. Out of a number of water chemistry metrics, elevated electrical conductivity (EC) was most strongly correlated with altered macroinvertebrate assemblages. They also identified 300–500 $\mu S\ cm^{-1}$ as a threshold above which sensitive macroinvertebrates were impacted. Additional research continues to identify strong correlations between mining and water quality impairment (e.g., elevated EC, $SO_4$, and Se) (Lindberg et al. 2011). Also, researchers continue to implicate mine drainage in declines of sensitive macroinvertebrate taxa in eastern Kentucky, USA, headwater streams (Pond 2010, 2011) and altered biofilm function/development in southeast Ohio, USA, (Smucker and Vis 2011). Finally, Hopkins et al. (2013) confirmed that these effects (elevated EC, $SO_4$, and Al) can be observed even in streams draining legacy sites long after conventional reclamation (>10 years).

With altered streamflow and water quality in mind, mitigation of the effect of mining by appropriate mine reclamation practices is critical. Optimal reclamation strategies should focus on restoring original terrestrial ecosystem structure and function as well as mitigating the impacts of mining on water quality and hydrology. Currently, surface mine reclamation is governed by the Surface Mine Control and Reclamation Act (SMCRA) of 1977. This legislation was enacted in large part to correct the poor reclamation practices of the day, which were known to contribute to flooding, sedimentation, and slope instability. Thus, SMCRA focused on mandating erosion control and slope stability. Over time, however, interpretation of SMCRA led operators to heavily compact mine soils and seed with competitive grasses and legumes to quickly achieve high vegetative cover. In addition, a poorly defined clause allowing the use of the best available soil substitute in place of native topsoil led to the use of crushed unweathered overburden as a growth medium. Thus, post-SMCRA-reclaimed sites are characterized by heavily compacted alkaline soils, which are unfavorable to native hardwood trees and tend to remain in arrested succession, covered by competitive nonnative grasses and legumes (Angel et al. 2005).

In an effort to develop surface mine reclamation techniques that help encourage mined systems toward native ecosystem structure and function, the Appalachian Regional Reforestation Initiative (ARRI) was formed. The ARRI science team has subsequently published a series of advisories detailing the Forestry Reclamation Approach (FRA): (1) select suitable material for a growth medium, (2) minimize compaction, (3) minimize competition from groundcover, (4) plant early- and late-succession tree species, and (5) use proper planting techniques (Zipper et al. 2011). The first of these points, growth medium selection, addresses what has been a critical knowledge gap in Appalachian surface mine reclamation: defining the best available soil substitute.

A number of studies were designed to compare weathered and unweathered spoils as soil substitutes in reclamation; all of them have concluded that brown, weathered sandstones provide a more favorable growth medium for native hardwood tree species than gray, unweathered spoils (Angel 2008; Emerson et al. 2009; Showalter et al. 2010; Agouridis et al. 2012; Miller et al. 2012; Wilson-Kokes et al. 2013a, b). However, the influence of spoil type on water quality remains an important research need. In a study comparing spoil types on Bent Mountain in eastern KY (2005–2007), Agouridis et al. (2012) reported that brown weathered sandstone discharged water with lower EC than both gray unweathered sandstone and mixed sandstone and shale spoil, although all spoil types discharged water with average EC levels greater than 500 $\mu S\ cm^{-1}$ (829; 1,032; and 1,224, respectively). They also observed that most chemical constituents were characterized by negative trends, suggesting that initially high levels are temporary.

Also, little is known regarding the influence of spoil type on water retention and discharge during afforestation on mined sites. On the Bent Mountain study site, Taylor et al. (2009a) found no difference in runoff curve numbers between end-dumped loose spoils and a reference undisturbed forest. These results suggest that the minimal compaction required by FRA step 2 may be sufficient to address some of the runoff volume issues associated with reclaimed mine sites (Taylor et al. 2009a). During the same study, Taylor et al. (2009b) reported that discharge volume did not differ across spoil types (BROWN, GRAY, and MIXED); however, with tree growth higher on BROWN by the third study year (2007), the authors suggested that continued forest development would lead to increased interception and storage, thus reducing discharge volumes (Taylor et al. 2009b).

This study was designed to follow-up on the previous work by Agouridis et al. (2012), Angel (2008), and Taylor et al. (2009a, b). Similar studies (e.g., Wilson-Kokes et al. 2013a, b) have reported 8-year data on experimental FRA spoil segregation plantings; however, these studies have primarily focused on tree growth. This paper will present 9-year data on the experimental spoil segregation FRA planting on Bent Mountain, emphasizing the influence of spoil type on water chemical and hydrological profiles. We hypothesize that quality of water discharged from the plots will vary among treatments. Because BROWN has been previously observed to be more favorable to tree growth (Angel 2008) and to discharge higher quality water than GRAY or MIXED (Agouridis et al. 2012), we predict that BROWN will have the highest quality water. We also predict that 2012–2013 water quality will be improved from 2005 to 2007 water quality, given the trends observed by Agouridis et al. (2012). In addition, because of the disparity among treatments in tree growth and vegetative colonization observed in 2007 (Angel 2008), we hypothesize that continued forest development on these sites would have led to reduced discharge volumes from BROWN compared to GRAY or MIXED, especially during the growing season when evapotranspiration is a stronger influence.

## 2 Methods

### 2.1 Plot Construction

Experimental plots were constructed on the Bent Mountain surface mine in Pikeville, KY, USA, in 2005 (Angel 2008). Three spoil type treatments (BROWN weathered sandstone, GRAY unweathered sandstone, and MIXED sandstone and shale) were end-dumped in 0.4-ha plots in duplicate for a total of six plots. Prior to end-dumping, a base layer was heavily compacted to prevent infiltration and graded to drain interflow via a drainage tile out to one side of each plot. Interflow passed through monitoring equipment (tipping bucket) and was drained into an underlying deep mine to effectively isolate the plots from each other. Here, interflow simulates water discharged to groundwater and/or streamflow in a natural system. After end-dumping, plots were graded with one pass of a small bulldozer to strike off the tops of the spoil piles and create a more level topography. Excessive grading was avoided to minimize compaction. After plot construction, four native hardwood species were planted directly into the spoil: green ash (*Fraxinus pennsylvanica*), red oak (*Quercus rubra*), white oak (*Quercus alba*), and yellow poplar (*Liriodendron tulipifera*). No groundcover was seeded on these plots to minimize vegetative competition. Tree survival and growth, unassisted groundcover colonization, water quality, and hydrology were monitored on these plots from 2005 to 2007 and reported in Angel (2008) and Agouridis et al. (2012).

### 2.2 Soil

Soil samples were collected in January 2013. Each plot ($n$=6) was subdivided into 16 subplots, 8 of which were randomly selected as subsampling locations. Within each subplot, six samples were collected using a sampling trowel and thoroughly mixed on site to give a composite sample representative of both ridge and depression areas. Thus, eight subsamples were collected per plot, for a total of 48 samples. Because these plots were extremely rocky, soil sampling was only 6–8 cm deep. Soil samples were analyzed by the University of Kentucky Regulatory Services Soils Laboratory. Sand, silt, and clay were determined by the micropipette method (Miller and Miller 1987); pH was determined in a 1:1 soil water solution (Thomas 1996). Soil EC was determined in a soil water extract (Soil and Plant Analysis Council 2000).

Means were calculated by plot for each soil analysis parameter, and these were analyzed by ANOVA using PROC MIXED (SAS 9.3), with treatment modeled as fixed effect and rep ($n$=2) as random effect. Year was modeled in the repeated statement. Significant ANOVAs ($p$<0.05) were followed up with pairwise comparisons of least-squared means.

### 2.3 Trees and Groundcover

Tree and groundcover data were collected in August 2013. Tree height was measured with a telescoping measuring rod, and tree diameter was measured with calipers. Tree volume index (TVI) was calculated according to the formula TVI=$d^2h$. The number of surviving planted trees was recorded. Unassisted vegetative colonization was evaluated according to the method of Farmer et al. (1981) for measuring groundcover on reclaimed surface mines. Percent ground cover and species composition were recorded.

### 2.4 Water Quality

As interflow drained via a drainage tile out of each plot, it passed through a data acquisition station equipped with a tipping bucket (to record flow information), and a ~5 mL aliquot was collected from every other tip in a collection bottle. During sampling trips, this flow-weighted sample was collected and transported to the UK Department of Forestry Hydrology Lab for analysis. Water pH and alkalinity were measured using an autotitrater. EC was measured using a YSI conductivity bridge. Ca, K, Mg, and Na, Fe, and Mn were measured using a GBC SDS 270 Atomic Adsorption Spectrophotometer (GBC Scientific Equipment, Melbourne, Australia). $NH_4$-N and $NO_3$-N were measured by colorimetric analysis using a Bran + Luebbe autoanalyzer (Bran + Luebbe, Analyzer Division, Germany). $SO_4$, $PO_4$, and Cl were measured using ion chromatography on a Dionex Ion Chromatograph 2000 (Dionex Corp., CA). Finally, a subset of our samples were analyzed at the University of Kentucky Environmental Research Training Laboratories (ERTL) against a multi-elemental standard (Al, As, Ca, Cd, Cr, Cu, Fe, Ni, Pb, and Zn) by inductively coupled plasma-optimal emission spectroscopy (ICP-OES)(Varian Vista-Pro CCD Simultaneous, Palo Alto, CA).

Water quality data were analyzed using PROC MIXED (SAS 9.3), with treatment, days (since first sample in current study), and treatment*days interaction modeled as fixed effects and rep ($n$=2) as a random effect. The spatial power SP (POW) covariance structure provided the best fit for uneven repeated measures sampling intervals. Linear regressions were performed on all data from the current sampling period and compared with regressions of 2005–2007 data (Agouridis et al. 2012).

### 2.5 Hydrology

Tipping bucket data were downloaded during regular sampling trips. Flow data were categorized as growing/dormant season according to the growing season for Pike County (NRCS Soil Survey). Due to early observations of impaired hydrologic function (interflow was discharged only briefly during extreme storm events) on one of the GRAY plots (Angel 2008), discharge data were only analyzed for one replicate of GRAY. Due to equipment malfunction on GRAY and MIXED plots, accurate flow data were not recorded for some periods. To correct for this, only data from periods consistent across all plots were analyzed. For each plot, total period flow (including both stormflow and baseflow) was summed and expressed as a percentage of total precipitation during that period. Storm response on GRAY and MIXED tended to max out dataloggers, resulting in incomplete captures of storm hydrographs. Because of this, our data correction methods should result in an underestimation of total flow on GRAY and MIXED and a relative overestimation of flow from BROWN, making our conclusions conservative.

Rainfall data were obtained from the nearby USGS gage station in Meta, KY (Gage #03210000). Total rainfall was summed according to periods of available flow data. Discharge was expressed as a percentage of incident precipitation. Hydrology data were analyzed by PROC MIXED, with season (growing 2012, dormant 2012–2013, and growing 2013), treatment, and season*treatment interaction as fixed effects and rep as random effect.

## 3 Results and Discussion

### 3.1 Soil

Soil physical and chemical characteristics were more favorable for native plant growth on BROWN than either MIXED or GRAY (Table 1). Soil particle size distribution is one of the major differences among spoil types, with clay fraction greatest on BROWN (10.3 %), followed by MIXED (9.1 %) and GRAY (7 %). Clays are the smallest soil particle size class and provide important soil water and cation holding functions. While mean percent clays were greatest on BROWN, mean percent silts were greatest on MIXED and rapidly increasing—consistent with observations by Miller et al. (2012) that spoils containing shales weather into fines more rapidly than spoils without shale. We observed greater field capacity on BROWN than MIXED or GRAY, consistent with observations by Angel (2008) that soil moisture was greater on BROWN than MIXED or GRAY. As the fine soil fraction continues to increase on BROWN and MIXED, we would expect a similar increase in water holding capacity leading to increasing favorability for native plant growth.

Soil on BROWN is also chemically more favorable to plant growth. Since plot construction in 2005, soil pH on BROWN has been most similar to native Appalachian soils, ranging from 6 to 6.5. In contrast, soil pH on GRAY and MIXED was greater than 8.0 since construction, outside the growing range of most native hardwoods (USDA 1973; Williston and LaFayette 1978). Soil pH rose in all spoils from 2005 to 2007, which was attributed to rapid weathering of carbonates. Carbonate minerals react in the presence of free protons ($H^+$) to give alkaline bicarbonate ions which raise pH in both soil and soil solution (Shrestha and Lal 2007; Maharaj et al. 2007). Another important trend in soil chemistry is EC. Soil EC decreased drastically from 2005 (163–185 $\mu S\ cm^{-1}$) to 2013 (73–87 $\mu S\ cm^{-1}$). Initially high soil EC indicated that the pool of readily soluble salts was high; this was mirrored in high EC in interflow discharged from the plots (Agouridis et al. 2012). Decreased soil EC suggests that this pool is declining.

### 3.2 Trees and Groundcover

While initial tree volume was similar across treatments (i.e., planted trees were essentially the same size), Angel (2008) observed significantly greater growth on BROWN than MIXED or GRAY by 2007. This trend continued in 2013, with tree volume on BROWN nearly 50 times greater than GRAY and nearly seven times greater than MIXED (Table 2). Consistent with predictions from soil data, it is clear that BROWN provides a

**Table 1** Means and standard errors of soil pH, electrical conductivity, and texture classes

| Parameter | 2005 | | | 2013 | | |
|---|---|---|---|---|---|---|
| | Brown | Gray | Mixed | Brown | Gray | Mixed |
| pH | 6.03b±0.47 | 8.07 a±0.13 | 8.33a±0.01 | 6.07b±0.46 | 8.44a±0.29 | 8.12a±0.08 |
| EC (μS $cm^{-1}$) | 163a±21 | 160a±1.6 | 185a±6.5 | 73a±8.1 | 78a±21 | 86a±0.6 |
| % Sand | 60.8b±0.63 | 77.8a±0.66 | 73.9b±2.84 | 64.1b±1.69 | 74.2a±0.73 | 57.4b±3.68 |
| % Silt | 27.2a±0.63 | 15.7b±0.35 | 18.4a±1.76 | 25.6a±0.611 | 18.8b±0.43 | 33.4a±3.26 |
| % Clay | 11.9a±1.25 | 6.5b±0.32 | 7.7b±1.1 | 10.3a±1.08 | 7.0b±0.30 | 9.1b±0.42 |

Means with the same letter are not significantly different ($p<0.05$) across treatment within years

more suitable growth medium for native hardwoods than either GRAY or MIXED and that this trend continues through nine growing seasons. Also, worth noting is that tree volume on MIXED is significantly greater than on GRAY, suggesting that spoil with some weathered material incorporated may provide a better rooting medium than unweathered spoil alone.

Angel (2008) reported that tree survival through 2007 was higher on GRAY than on MIXED or BROWN. This was somewhat anomalous, because it was clear at that point that tree volume was higher on BROWN than on the other spoils. By 2013, however, survival on GRAY and MIXED had dropped below 70 %, while survival on BROWN stayed at 86 % (Table 2). Given the soil quality issues addressed above, it is likely that cumulative stresses associated with unfavorable soil pH and poor soil moisture were responsible for the decline in survival on GRAY and MIXED. This decline in survival was most obvious in white oak, which dropped from >90 % on GRAY and MIXED in 2007 to <50 % in 2013. Soil pH on GRAY and MIXED is outside the pH range of native soils in which white oak thrive (Cotton 2006).

In addition to the survival and growth of planted trees, we characterized the species richness and cover of a naturally colonized understory community (Table 3). During plot construction, consistent with FRA recommendation 3 (minimize vegetative competition), no groundcover was seeded on the sites. However,

**Table 2** Tree volume index ($cm^3$) and percent survival, with standard errors, by spoil type and year

| Volume | 2005 | | | 2007 | | | 2013 | | |
|---|---|---|---|---|---|---|---|---|---|
| | Brown | Gray | Mixed | Brown | Gray | Mixed | Brown | Gray | Mixed |
| GA | 11.6a±1.9 | 14.6a±0.2 | 13.4a±0.6 | 238a±100 | 41.9a±1.1 | 99.6a±20.4 | 8,007a±894 | 302c±105 | 1,420b±11.3 |
| RO | 13.35a±1.0 | 14a±0.7 | 13.0a±1.0 | 179a±27 | 26.1a±5.3 | 65.6a±5.6 | 10,524a±993 | 162c±127 | 1,399b±87 |
| WO | 8.4a±1.4 | 6.8a±0.2 | 7.8a±0.2 | 94a±23 | 8.8a±2.2 | 14.3a±3.6 | 8,982a±1,889 | 68b±56 | 134b±72 |
| YP | 29.7a±0.8 | 30.0a±1.0 | 26.3a±3.1 | 440a±157 | 72.2a±15.2 | 179a±19.2 | 25,241a±2,409 | 380c±147 | 4,183b±1,280 |
| All | 16.1a±1.1 | 16.2a±1.0 | 15.2a±0.4 | 238a±76 | 34.8a±6.9 | 85.7a±10.3 | 12,270a±292 | 237c±115 | 1,837b±277 |
| Survival (%) | | | | | | | | | |
| GA | – | | | 96.9±0.00 | 96.1±0.00 | 95.6±0.04 | 94.1±0.01 | 90.6±0.02 | 97.0±0.02 |
| RO | – | | | 77.1±0.02 | 79.9±0.01 | 72.9±0.02 | 78.1±0.02 | 74.3±0.09 | 66.9±0.04 |
| WO | – | | | 100±0.11 | 100±0.12 | 92.6±0.12 | 100±0.16 | 100±0.22 | 48.0±0.11 |
| YP | – | | | 69.9±0.09 | 72.5±0.06 | 65.4±0.00 | 65.3±0.08 | 65.3±0.22 | 62.3±0.05 |
| All | – | | | 87.0±0.00 | 87.8±0.03 | 81.3±0.02 | 86.4±0.02 | 86.4±0.15 | 67.9±0.05 |

Means with the same letters are not significantly different ($p=0.05$) among spoil types within year

*GA* green ash, *RO* red oak, *WO* white oak, *YP* yellow poplar, *All* all planted species

**Table 3** Mean percent groundcover and species richness (with standard errors) provided by volunteer vegetation by spoil type and year

| Year | | Brown | Gray | Mixed |
|---|---|---|---|---|
| 2006 | Cover | 42.3±0.03 | 1.0±0.0 | 2.6±0.00 |
| | N | 40±1 | 6±0.5 | 21±2 |
| 2007 | Cover | 66.4±0.06 | 2.0±0.00 | 5.8±0.00 |
| | N | 61±8 | 12±0.5 | 35±0.5 |
| 2013 | Cover | 99.1±0.00 | 9.8±0.05 | 20.2±0.07 |
| | N | 57±5 | 42±4.5 | 43±0.5 |

by 2006 after planting, Angel observed that naturally colonizing vegetation provided greater groundcover on BROWN than either MIXED or GRAY. In 2013, this trend was similar to the trend observed in tree growth, with groundcover on BROWN at nearly 100 % while GRAY and MIXED were much lower (9.8 and 20.2 %, respectively). In addition to consistently higher percent groundcover, species richness was higher in BROWN than in GRAY or MIXED all 3 years sampled. Thus, BROWN clearly provided a more favorable medium for establishment of a range of naturally colonizing species.

Average volume of planted tree species on BROWN in 2013 was 50 times greater than tree volume on BROWN in 2007. With this accelerated growth, significant forest structure changes have been observed. For instance, planted and volunteer trees are attaining canopy closure in some areas. This shading has led to localized extirpation of invasive Chinese lespedeza (*Lespedeza cuneata*) and has favored the colonization of shade-tolerant native species like wild hydrangea (*Hydrangea* sp.) and alumroot (*Heuchera* sp.). The development of shade-tolerant understory communities demonstrates progression of BROWN toward a system more similar to native Appalachian forest (Hall et al. 2010).

## 3.3 Water Quality

Since water with high EC was implicated as a potential driver of macroinvertebrate assemblage shifts (Pond et al. 2008), developing reclamation techniques that control EC has been a priority. EC was significantly different among treatments through the first 3 years (BROWN, 829 μS $cm^{-1}$; GRAY, 1,032 μS $cm^{-1}$; MIXED, 1,224 μS $cm^{-1}$); however, no significant differences were detected in 2012–2013 samples (BROWN, 421 μS $cm^{-1}$; GRAY, 564 μS $cm^{-1}$; MIXED, 455 μS $cm^{-1}$) (Table 4). Agouridis et al. (2012) found that EC, while initially much higher in MIXED and GRAY, declined more steeply in these spoils than in BROWN and projected that they would be below the proposed 500 μS $cm^{-1}$ threshold by 2008. However, this rapid rate of leaching and weathering did not persist. Regression analysis of 2012–2013 data revealed no significant trends in EC (i.e., the slope of the regression line is not different from zero, Table 5). Data from 2005 to 2007 suggest that weathering and leaching processes occur rapidly during the first few years after mine spoil placement (Fig. 1). EC did not decrease indefinitely; rather, we conclude from 2012 to 2013 data that EC reached an equilibrium state in which input from weathering minerals was roughly equal to output from leaching via discharged interflow.

**Table 4** Means and standard errors of constituent concentration of water samples by spoil type during the sample period (March 2012 through November 2013)

| Parameter | BROWN | GRAY | MIXED |
|---|---|---|---|
| EC (μS $cm^{-1}$) | 421±39 | 564±55 | 455±39 |
| Cl (mg $L^{-1}$) | 1.4±0.2 | 1.2±0.3 | 0.8±0.2 |
| $SO_4$ (mg $L^{-1}$) | 66.6±8.4 | 30.6±11.8 | 29.9±8.3 |
| Mg (mg $L^{-1}$) | 11.2±0.8 | 13.5±1.1 | 12.9±0.8 |
| Ca (mg $L^{-1}$) | 23.4±1.3 | 14.0±1.8 | 19.8±1.3 |
| K (mg $L^{-1}$) | 3.6±0.4 | 8.5±0.5 | 7.2±0.4 |
| Na (mg $L^{-1}$) | 5.4±1.5 | 1.2±2.1 | 1.6±1.4 |
| Alkalinity (mg $HCO_3^{-}L^{-1}$) | 294±63.7 | 553±89.9 | 431±63.6 |
| pH | 7.3±0.1 | 7.9±0.2 | 7.8±0.1 |
| $NO_3$-N (mg $L^{-1}$) | 0.2±0.1 | 0.5±0.1 | 0.2±0.1 |
| $NH_4$-N (mg $L^{-1}$) | 0.1±0.0 | BDL | BDL |
| TOC (mg $L^{-1}$) | 53.8±13.2 | 100.8±18.5 | 67.1±13.1 |
| Fe (mg $L^{-1}$) | 0.4±0.2 | 0.0±0.3 | 0.0±0.2 |
| Mn (mg $L^{-1}$) | BDL | BDL | BDL |
| $PO_4$ (mg $L^{-1}$) | 0.2±1.2 | 2.8±1.8 | 1.8±1.2 |
| Zn (mg $L^{-1}$) | BDL | BDL | BDL |
| Al (mg $L^{-1}$) | 0.1±0.1 | 0.0±0.1 | 0.0±0.1 |
| As (mg $L^{-1}$) | BDL | BDL | BDL |
| Cd (mg $L^{-1}$) | BDL | BDL | BDL |
| Cr (mg $L^{-1}$) | BDL | BDL | BDL |
| Cu (mg $L^{-1}$) | BDL | BDL | BDL |
| Ni (mg $L^{-1}$) | BDL | BDL | BDL |
| Pb (mg $L^{-1}$) | BDL | BDL | BDL |

We found no significant treatment effects for any parameter ($p$=0.05)

*BDL* below detectable limit

**Table 5** Coefficients and intercepts (with standard errors) of constituent concentrations in water samples by spoil type (March 2012 through November 2013)

| Constituent | Treatment | | | | | |
|---|---|---|---|---|---|---|
| | Brown | | GRAY | | MIXED | |
| | Intercept | Slope | Intercept | Slope | Intercept | Slope |
| EC | 423.3±40.6 | −0.009±0.121 | 585.8±32.5 | −0.082±0.098 | 470.5±18.0 | −0.062±0.054 |
| Cl | 1.559±0.358 | 0.000±0.001 | 1.308±1.165 | 0.000±0.004 | 0.681±0.220 | 0.000±0.000 |
| $SO_4$ | 92.7±13.0 | −0.088±0.038 | 33.4±5.65 | −0.010±0.017 | 42.8±2.20 | −0.044±0.007 |
| Mg | 9.201±0.842 | 0.008±0.002 | 13.04±1.09 | 0.004±0.003 | 12.6±0.755 | 0.003±0.002 |
| Ca | 25.9±1.96 | −0.012±0.006 | 14.4±1.354 | −0.004±0.004 | 20.58±1.01 | −0.006±0.003 |
| K | 3.185±0.36 | 0.001±0.001 | 7.925±0.609 | 0.001±0.002 | 6.618±0.342 | 0.001±0.001 |
| Na | 5.91±0.748 | −0.001±0.002 | 1.133±0.120 | 0.000±0.000 | 1.522±0.091 | 0.000±0.000 |
| Alkalinity | 343.3±37.8 | −0.184±0.110 | 686.0±32.6 | −0.48±0.098 | 554.7±17.9 | −0.454±0.053 |
| pH | 7.334±0.013 | 0.000±0.000 | 8.052±0.118 | 0.000±0.000 | 7.775±0.084 | 0.000±0.000 |
| $NO_3$-N | 0.353±0.122 | 0.000±0.000 | 0.49±0.122 | 0.000±0.000 | 0.302±0.071 | 0.000±0.000 |
| $NH_4$-N | 0.104±0.023 | 0.000±0.000 | 0.039±0.017 | 0.000±0.000 | 0.071±0.013 | 0.000±0.000 |
| TOC | 51.3±10.4 | 0.010±0.03 | 99.5±16.0 | 0.005±0.048 | 68.7±7.92 | −0.006±0.024 |
| Fe | 0.552±0.213 | 0.000±0.001 | 0.0347±0.031 | 0.000±0.000 | 0.0451±0.0173 | 0.000±0.000 |
| Mn | 0.069±0.019 | 0.000±0.000 | 0.0346±0.018 | 0.000±0.000 | 0.043±0.017 | 0.000±0.000 |
| $PO_4$ | 0.236±0.332 | 0.000±0.001 | 2.79±1.72 | 0.000±0.004 | 0.916±2.043 | 0.0026±0.0054 |
| Zn | 0.06±0.05 | 0.000±0.000 | 0.081±0.059 | 0.000±0.000 | 0.022±0.0037 | 0.000±0.000 |
| Al | 0.353±0.079 | 0.001±0.000 | 0.0137±0.0052 | 0.000±0.000 | 0.0111±0.0032 | 0.000±0.000 |

Independent variable is days from first sample in current period. Trends are not significant if zero falls within two standard errors of the slope of the regression line

Sulfate has been identified in previous studies (Kennedy et al. 2003; Lindberg et al. 2011; Hopkins et al. 2013) as a major contributor to elevated EC and associated macroinvertebrate toxicity downstream of mining operations. Thus, improving reclamation procedures to reduce $SO_4$ levels is also of particular concern. Agouridis et al. (2012) observed that sulfates were increasing slightly in BROWN and proposed that this trend was due to the oxidation of pyrite. While sulfide oxidation is acid-forming, it is likely that pH would be buffered by the high-carbonate containing spoils, giving the neutral-alkaline pH observed in interflow from BROWN. Sulfate levels were still higher in BROWN (66.6 mg $L^{-1}$) than MIXED (29.9 mg $L^{-1}$) and GRAY (30.6 mg $L^{-1}$) in 2013, but concentrations in all plots are much lower than 2005–2007 levels (Fig. 2). In 2012–2013 samples, $SO_4$ levels declined slightly in BROWN. Periodic spikes in $SO_4$ suggest that additional deposits of unweathered material remain isolated from interflow. As expanding root systems open new flowpaths, we anticipate that the number of such isolated pockets will continue to decrease. This should correspond to a decline in sulfates as well as EC.

Alkalinity declined in all treatments but was lower in BROWN (294 mg $L^{-1}$) than in GRAY (553 mg $L^{-1}$) or MIXED (431 mg $L^{-1}$) (Fig. 3). Declining alkalinity suggests that much of the rapidly dissolvable carbonate containing minerals has been weathered, leaving a more stable mineral fraction. This observation is consistent with similar trends in $SO_4$ and EC, which collectively suggest that the highly soluble/reactive mineral fraction in these plots rapidly declined from 2005 to 2007 and was equilibrated or nearing equilibration by 2013.

From 2005 to 2006, pH was distinctly increasing across treatments, from 7.5 to 8.5 (Agouridis et al. 2012). This was attributed primarily to the weathering of carbonate-containing minerals. While no significant differences were observed among treatments in the current sampling period, mean pH of water from BROWN (7.3) was lower than that from GRAY (7.9) or MIXED (7.8), all lower than mean pH values in 2007 (Fig. 4). These results are consistent with decreasing alkalinity in suggesting that

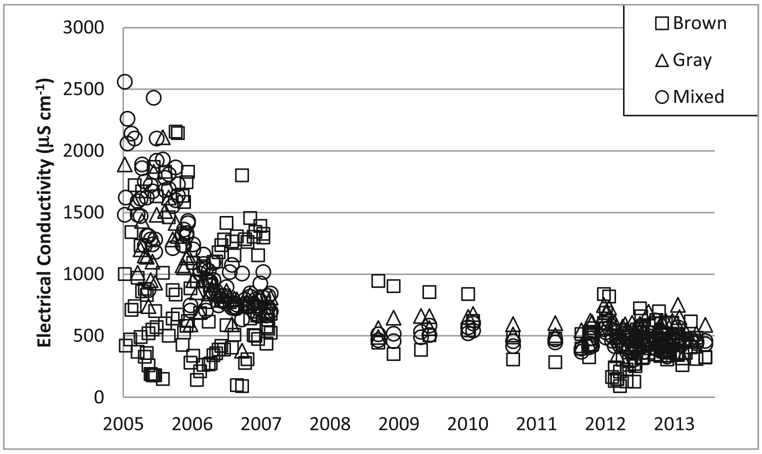


**Fig. 1** Electrical conductivity of flow-weighted samples, separated into sampling groups (2005–2007) and (2009–2013)

the initial phase of rapid carbonate leaching is over, and a more steady state system is in place.

Also of interest, pH in interflow from BROWN is slightly alkaline, while soil pH is slightly acidic. Ward (2009) observed that carbonate leaching was occurring more rapidly on BROWN than on MIXED or GRAY. Soil acidity (including acidity generated by oxidation of potential pyritic deposits) has contributed to rapid weathering and leaching of buffering carbonate compounds. As carbonates are leached from the system, soil pH will continue to decline. It is likely that continued plant growth will accelerate this process through the contribution of carbonic acid sourced in $CO_2$ produced by root cellular respiration and the opening up of new flowpaths (Maharaj et al. 2007).

No apparent trends were observed in concentrations of Mg, K, Na, or Cl. Nominally higher Ca concentration in BROWN may be indicative of continued carbonate leaching from these plots. Other than Al, trace metals (Mn, Zn, As, Cd, Cr, Cu, Ni, and Pb) were below the detectable limits of our procedure.

Although spoil type significantly influenced discharged water quality in 2005–2007 (Agouridis et al. 2012), we found that water quality was similar across spoil types in 2012–2013. Overall, we found that water quality in the 2012–2013 period was much improved compared to samples collected 2005–2007. High concentrations of metals and salts in 2005–2007 samples indicated that freshly placed mine spoil provides a pool of highly reactive species which can be readily dissolved by rainwater, leached through the profile, and discharged downstream. In mining operations, this leachate is responsible for downstream water pollution and macroinvertebrate community impairment

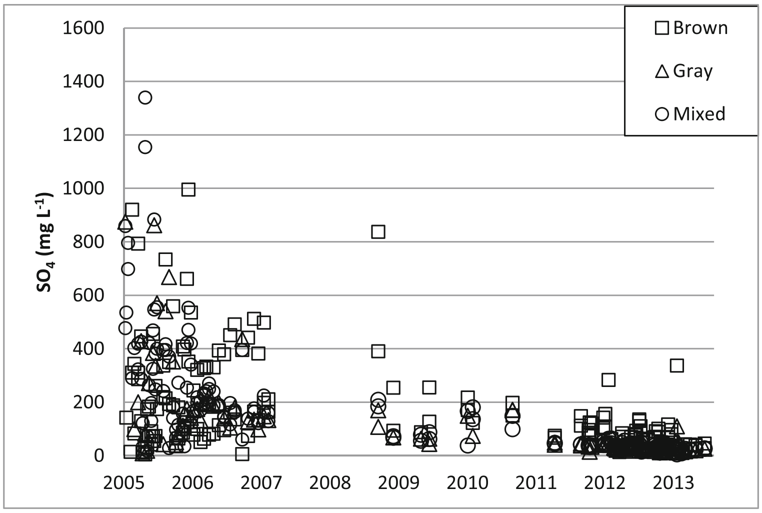


**Fig. 2** Sulfate concentrations in flow-weighted samples, separated into sampling periods 2005–2007 and 2009–2013

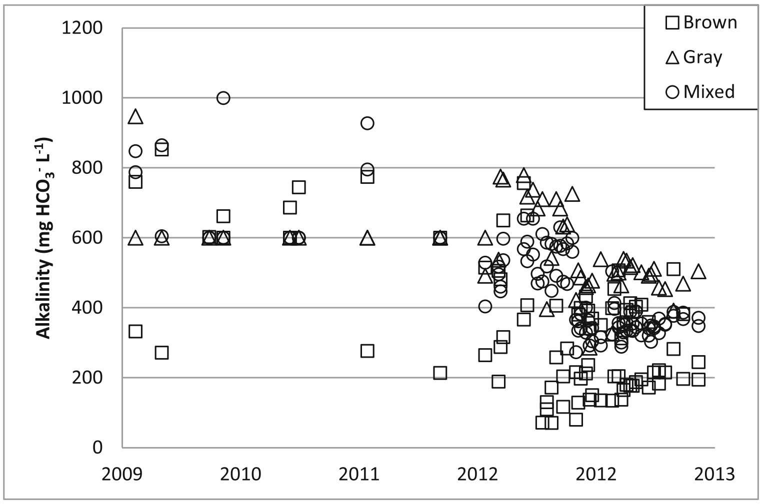


**Fig. 3** Alkalinity (as mg $HCO_3^-$ $L^{-1}$) of flow-weighted samples, 2009–2013

(Lindberg et al. 2011; Pond et al. 2008). In our study, although initial concentrations of most chemical species were quite high, Agouridis et al. (2012) found that most concentrations declined rapidly during the 2005–2007 period. By 2012–2013, however, slopes of regression lines for most constituents were not significantly different from zero, indicating that concentrations of most constituents had essentially leveled off. We conclude that mine spoils placed according to the FRA recommendations present a highly reactive pool of readily dissolvable salts. However, our data indicate that this pool can be substantially reduced in as few as 3–9 years after spoil placement. For example, mean EC in 2012–2013 was much lower than values from 2005 to 2007 and was within the biologically safe range identified by Pond et al. (2008). In contrast, conventionally reclaimed sites have been known to discharge water with EC as high as 2,000 μS/cm as late as 10 years after construction. These data provide strong support for utilization of FRA methods for spoil placement, with respect to mitigating some of the impacts of surface mining on aquatic ecosystems.

### 3.4 Hydrology

A major influence of surface mining on aquatic ecosystems has traditionally been hydrological alteration caused by heavy soil compaction leading to drastically reduced infiltration. Taylor et al. (2009a) observed that storm response on all treatments at Bent Mountain was similar to storm response in undisturbed reference forest. These observations suggest that minimizing

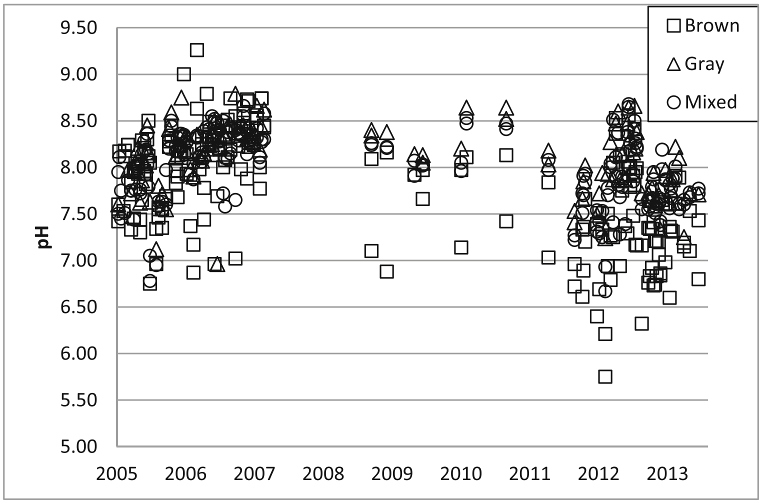


**Fig. 4** pH of flow-weighted samples, separated into sampling periods 2005–2007 and 2009–2013

compaction during spoil placement by end-dumping may reduce runoff volume during storm events relative to heavily compacted, traditionally reclaimed systems; further research is needed to compare an FRA system to a traditionally reclaimed system at a field scale. Through 2007, Taylor et al. (2009b) found only minor differences in discharge volume among plots but predicted that continued forest development would contribute to reduced discharge by improving soil water holding capacity and water interception by vegetation.

Forest cover is a critical component of the natural hydrologic function of the Appalachian region. For instance, one study found that over 19 % of incident precipitation was lost by canopy interception and subsequent evaporation (Carlyle-Moses and Price 1999). Litter interception by itself accounted for an additional 2–5 % precipitation loss in Appalachian forest (Helvey and Patric 1965). Total precipitation loss to ET may be as high as 43–45 % (Swank et al. 2001). Given that such a significant portion of the regional water budget is loss via ET, it is unsurprising that the extensive forest loss caused by surface mining would lead to subsequent loss of ET and increase in streamflow (Dickens et al. 1989; Messinger and Paybins 2003). Similarly, restoration of forest after surface mining would be expected to at least partially restore ET and rescue natural water budgeting.

By 2013, seasonal variation in discharge volume within treatment and across seasons was observed (Fig. 5). Discharge was most highly variable on BROWN, which had significantly lower discharge during the 2012 and 2013 growing seasons than the intermediate dormant season. Discharge on MIXED was lower during the 2012 growing season than the subsequent growing season; however, the 2013 growing season was not different from the 2012–2013 dormant season. In contrast, discharge on GRAY was similar across growing and dormant seasons. In addition, we observed variation in discharge volume within season and across treatments. While discharge during the dormant season was similar across treatments (ranging from 31 % on BROWN to 43 % on GRAY and MIXED), discharge from BROWN was significantly less than GRAY during both 2012 and 2013 growing seasons (2.5 and 11 % on BROWN compared to 19 and 31 % on GRAY). Discharge on MIXED was similar to both BROWN and GRAY during the 2012 and 2013 growing seasons.

We propose that the volume of transpired water was significantly greater on BROWN than on GRAY. Because so much water was withdrawn via transpiration, a lower volume of water was discharged from the BROWN plots than from GRAY. This study demonstrated that transpiration demand from a developing forest can lead to a reduction in water discharge from a surface mine. If appropriate reforestation techniques (e.g., FRA) are implemented and weathered spoils are

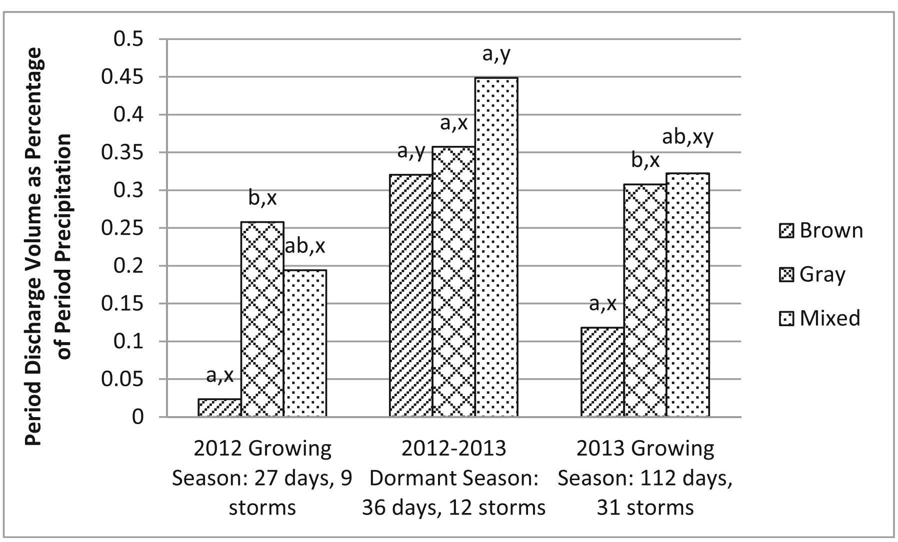


**Fig. 5** Discharge was summed for each plot during periods for which accurate data were available for all plots. Discharge volume was expressed as a percentage of precipitation volume. Number of days and storms indicate the number of days and storms for which accurate data were available during each respective season. *Bars with the same letter* (*abc*) are not significantly different among treatments within season. *Bars with the same letter* (*xyz*) are not significantly different across seasons within treatment

used in place of unweathered spoils when soil substitutes are required, forest development and subsequent transpiration effects may accelerate. As forest on BROWN continues to develop into the future, with related increases in transpiration, we anticipate that discharge from BROWN during the growing season will continue to decline. Because traditionally reclaimed mine sites may continue to discharge low-quality water (e.g., high EC, and high sulfates) for over a decade (e.g., Lindberg et al. 2011), reducing discharge volume may be a highly desirable feature for long-term downstream water quality recovery. Our results indicate that forest development on FRA-reclaimed sites may effectively reduce discharge volume only nine growing seasons after planting; thus, reclamation by FRA may accelerate water quality recovery after mining compared to traditional reclamation. Additional research into this relationship should be conducted at the watershed scale to investigate the potential for reforestation according to FRA techniques to reduce the influence of the mining process on downstream aquatic communities.

## 4 Conclusions

After nine growing seasons, it appears that BROWN unweathered spoils provide an improved growth medium for native trees and may indirectly contribute to accelerated water quality recovery from mining. Overall plant preference for BROWN was demonstrated by significantly higher tree volume and groundcover in BROWN than in the other spoils. These results indicate that BROWN weathered spoils should be selected in place of more unweathered GRAY spoils when native topsoil is unavailable. Also of note, both groundcover and tree growth was higher in MIXED than in GRAY; from a vegetative perspective, MIXED spoil may provide an intermediate quality substrate that will develop more quickly into suitable growing medium than GRAY. It is difficult to project how the GRAY plots will continue to develop as a growth medium; follow-up studies are recommended to monitor long-term development.

Across spoil types, water chemistry drastically improved since the initial 2005–2007 sampling period. As indicated by soils data (reduced soil EC) and water chemistry data (EC, $SO_4$, alkalinity), the pool of readily soluble minerals and salts was much smaller after nine growing seasons than during the first 3 years after plot construction. With many chemical constituents in stable concentrations (e.g., slope=0), we suggest that these plots have reached or nearly reached equilibrium after only nine growing seasons.

These results indicate that mine spoils placed according to the FRA recommendations can reach target levels for EC of discharged water (300–500 $\mu S\ cm^{-1}$, Pond et al. 2008) within nine growing seasons. Although we found that BROWN weathered spoil was much more favorable than GRAY or MIXED as a growth medium for native trees, quality of water discharged from our plots did not differ across treatments. We conclude that spoil type does not significantly influence the quality of discharged water nine growing seasons after placement. However, tree growth and understory colonization on BROWN have exerted a considerable influence on the fate of precipitation on site, resulting in increased transpiration and reduced discharge relative to GRAY or MIXED. We conclude that forest development on BROWN weathered spoils placed according to the FRA recommendations will eventually reduce discharge volume, potentially reducing the long-term impact of surface mining on downstream aquatic communities.

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## References

Agouridis, C. T., Angel, P. N., Taylor, T. J., Barton, C. D., Warner, R. C., Yu, X., & Wood, C. (2012). Water quality characteristics of discharge from reforested loose-dumped mine spoil in eastern Kentucky. *Journal of Environmental Quality, 41*, 454–468.

Angel, P. N. (2008). Forest establishment and water quality characteristics as influenced by spoil type on a loose graded surface mine in eastern Kentucky. Dissertation. University of Kentucky.

Angel, P., Davis, V., Burger, J., Graves, D., Zipper, C. (2005). The Appalachian Regional Reforestation Initiative. U.S. Office of Surface Mining. Forest Reclamation Advisory Number 1. 2 p. (available from http://arri.osmre.gov/).

Bernhardt, E. S., & Palmer, M. A. (2011). The environmental costs of mountaintop mining valley fill operations for aquatic

ecosystems of the Central Appalachians. *Annals of the New York Academy of Science, 1223*, 39–57.
Carlyle-Moses, D. E., & Price, A. G. (1999). An evaluation of the Gash interception model in a northern hardwood stand. *Journal of Hydrology, 214*, 103–110.
Cotton, C. A. (2006). Developing a method of site quality evaluation for *Quercus alba* and *Liriodendron tulipifera* in the eastern Kentucky coal fields. Thesis. University of Kentucky.
Dickens, P. S., Minear, R. A., & Tschantz, B. A. (1989). Hydrologic Alteration of Mountain Watersheds from surface mining. *Journal Water Pollution Control Federation, 61*, 1249–1260.
Emerson, P., Skousen, J., & Ziemkiewicz, P. (2009). Survival and growth of hardwoods in brown versus gray sandstone on a surface mine in West Virginia. *Journal of Environmental Quality, 38*, 1821–1829.
Farmer, R. E., Jr., Rennie, J. C., Scanlon, D. H., III, and Zarger, T. G. (1981). Technical guides on use of reference areas and technical standards for evaluating surface mine vegetation in OSM Regions I and II. US Department of the Interior, Knoxville, TN.
Ferrari, J. R., Lookingbill, T. R., McCormick, B., Townsend, P. A., & Eshleman, K. N. (2009). Surface mining and reclamation effects on flood response of watersheds in the central Appalachian Plateau region. *Water Resources Research, 45*, W04407.
Hall, S. L., Barton, C. D., & Baskin, C. C. (2010). Topsoil seed bank of an oak-hickory forest in eastern Kentucky as a restoration tool on surface mines. *Restoration Ecology, 18*, 834–842.
Helvey, J., & Patric, J. (1965). Canopy and litter interception of rainfall by hardwoods of eastern United States. *Water Resources Research, 1*, 193–206.
Hopkins, R. L., Altier, B. M., Haselman, D., Merry, A. D., & White, J. J. (2013). Exploring the legacy effects of surface coal mining on stream chemistry. *Hydrobiologia, 713*, 87–95.
Hornbeck, J. W., Pierce, R. S., & Federer, C. A. (1970). Streamflow changes after forest clearing in New England. *Water Resources Research, 6*, 1124–1132.
Kennedy, A. J., Cherry, D. S., & Currie, R. J. (2003). Field and laboratory assessment of a coal processing effluent in the Leading Creek Watershed, Meigs County, Ohio. *Archives of Environmental Contamination and Toxicology, 44*, 324–331.
Kochenderfer, J. N., Adams, M. B., Miller,G. W., and Helvey, J. D. (2007). Factors affecting large peakflows on Appalachian watersheds: lessons from the Fernow Experimental Forest, Res. Pap. NRS-3, 24 pp., US Dep. of Agric. For. Serv. North. Res. Stn., Newtown Square, Pa. (Available at http://www.nrs.fs.fed.us/pubs/2838)
Lindberg, T. T., Bernhardt, E. S., Bier, R., Helton, A. M., Merola, R. B., Vengosh, A., & Di Giulio, R. T. (2011). Cumulative impacts of mountaintop mining on an Appalachian watershed. *Proceedings of the National Academy of Science, 108*, 20929–20934.
Maharaj, S., Barton, C. D., Karathanasis, T. A. D., Rowe, H. D., & Rimmer, S. M. (2007). Distinguishing "new" from "old" organic carbon on reclaimed coal mine sites using thermogravimetry: I. Method development. *Soil Science, 172*, 292–301.
Messinger, T., & Paybins, K. S. (2003). *Relations between precipitation and daily and monthly mean flows in gaged, unmined and valley-filled watersheds, Ballard Fork, West Virginia, 1999-2001*. Charleston, WV: USGS.
Miller, W. P., & Miller, D. M. (1987). A micropipette method for soil mechanical analysis. *Communications in Soil Science and Plant Analysis, 18*, 1–15.
Miller, J., Barton, C., Agouridis, C., Fogel, A., Dowdy, T., & Angel, P. (2012). Evaluating soil genesis and reforestation success on a surface coal mine in Appalachia. *Soil Science Society of America Journal, 76*, 950–960.
Negley, T. L., & Eshleman, K. N. (2006). Comparison of stormflow responses of surface-mined and forested watersheds in the Appalachian Mountains, USA. *Hydrological Processes, 20*, 3467–3483.
Pond, G. J. (2010). Patterns of Ephemeroptera taxa loss in Appalachian headwater streams (Kentucky, USA). *Hydrobiologia, 641*, 185–201.
Pond, G. J. (2011). Biodiversity loss in Appalachian headwater streams (Kentucky, USA): Plecoptera and Trichoptera communities. *Hydrobiologia, 679*, 97–117.
Pond, G. J., Passmore, M. E., Borsuk, F. A., Reynolds, L., & Rose, C. J. (2008). Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. *Journal of the North American Benthological Society, 27*, 717–737.
Ritter, J. B., & Gardner, T. W. (1993). Hydrologic evolution of drainage basins disturbed by surface mining, central Pennsylvania. *Geological Society of America Bulletin, 105*, 101–115.
Schmidt, T. S., Soucek, D. J., & Cherry, D. S. (2002). Integrative assessment of benthic macroinvertebrate community impairment from metal-contaminated waters in tributaries of the Upper Powell River, Virginia, USA. *Environmental Toxicology and Chemistry, 21*, 2233–2241.
Showalter, J. M., Burger, J. A., & Zipper, C. E. (2010). Hardwood seedling growth on different mine spoil types with and without topsoil amendment. *Journal of Environmental Quality, 39*, 483–491.
Shrestha, R. K., & Lal, R. (2007). Soil carbon and nitrogen in 28-year-old land uses in reclaimed coal mine soils of Ohio. *Journal of Environmental Quality, 36*, 1775–1783.
Smucker, N. J., & Vis, M. L. (2011). Acid mine drainage affects the development and function of epilithic biofilms in streams. *Journal of the North American Benthological Society, 30*, 728–738.
Soil and Plant Analysis Council. (2000). Soil analysis handbook of reference methods. Boca Raton, FL: CRC Press
Soucek, D. J., Cherry, D. S., Currie, R. J., Latimer, H. A., & Trent, G. C. (2000). Laboratory to field validation in an integrative assessment of an acid mine drainage-impacted watershed. *Environmental Toxicology and Chemistry, 19*, 1036–1043.
Swank, W. T., Vose, J. M., & Elliott, K. J. (2001). Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a southern Appalachian catchment. *Forest Ecology and Management, 143*, 163–178.
Taylor, T. J., Agouridis, C. T., Warner, R. C., & Barton, C. D. (2009a). Runoff curve numbers for loose-dumped spoil in the Cumberland Plateau of eastern Kentucky. *International*

*Journal of Mining, Reclamation and Environment, 23*, 103–120.

Taylor, T. J., Agouridis, C. T., Warner, R. C., Barton, C. D., & Angel, P. N. (2009b). Hydrologic characteristics of Appalachian loose-dumped spoil in the Cumberland Plateau of eastern Kentucky. *Hydrological Processes, 23*, 3372–3381.

Thomas, G. W. (1996). Soil pH and soil acidity. In D. L. Sparks & J. M. Bartels (Eds.), *Methods of soil analysis. Part 3, chemical methods* (pp. 475–790). Madison, WI: ASA and SSSA.

[USDA] US Department of Agriculture. (1973). Important trees of eastern forests. U.S. Forest Service, Southern Region, Atlanta, GA.

Ward, K. M. (2009). Matrix geochemistry and *Phytophthora* occurrence on reforested mine lands in Appalachia. Thesis. University of Kentucky.

Williston, H. L., and LaFayette, R. (1978). Species suitability and pH of soils in southern forests. State and Private Forestry, USDA Forest Service Southeastern Area. 4 p.

Wilson-Kokes, L., DeLong, C., Thomas, C., Emerson, P., O'Dell, K., & Skousen, J. (2013a). Hardwood tree growth on amended mine soils in West Virginia. *Journal of Environmental Quality, 42*, 1363–1371.

Wilson-Kokes, L., Emerson, P., DeLong, C., Thomas, C., & Skousen, J. (2013b). Hardwood tree growth after eight years on brown and gray mine soils in West Virginia. *Journal of Environmental Quality, 42*, 1353–1362.

Zipper, C. E., Burger, J. A., Skousen, J. G., Angel, P. N., Barton, C. D., Davis, V., & Franklin, J. A. (2011). Restoring forests and associated ecosystem services on Appalachian coal surface mines. *Environmental Management, 47*, 751–765.