2014

Influence of Spoil Type on Afforestation Success and Hydrochemical Function on a Surface Coal Mine in Eastern Kentucky

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INFLUENCE OF SPOIL TYPE ON AFFORESTATION SUCCESS AND HYDROCHEMICAL FUNCTION ON A SURFACE COAL MINE IN EASTERN KENTUCKY

THESIS

A thesis submitted in partial fulfillment of the requirements for the degree of Master of Science in the College of Agriculture, Food, and the Environment at the University of Kentucky

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2014

ABSTRACT OF THESIS

INFLUENCE OF SPOIL TYPE ON AFFORESTATION SUCCESS AND HYDROCHEMICAL FUNCTION ON A SURFACE COAL MINE IN EASTERN KENTUCKY

Surface coal mining in Appalachia has contributed to a suite of ecological impacts, both terrestrial and aquatic. Conventional reclamation in Appalachia leads to the development of hay/pasture systems dominated by nonnative grasses and legumes, with soils that are chemically and physically unfavorable to native tree growth. Several studies have shown that more weathered minespoils provide a better growth medium than unweathered spoils in Appalachia. Spoil segregation plots were constructed on Bent Mountain in Pike County, KY, to compare the suitability of three mine spoil types (BROWN weathered sandstone, GRAY unweathered sandstone, and MIXED sandstones and shales). In 2013 (after nine growing seasons) volume of planted trees was 50x higher on BROWN than on GRAY. In addition, natural colonization of unplanted groundcover and tree species was much more extensive on BROWN than GRAY or MIXED. Most water chemical parameters were similar across spoil types; however, water chemistry on all plots appears to have stabilized after nine growing seasons. Finally, rapidly developing forest on BROWN appears to be influencing water budgeting on the site, leading to lower discharge during summer months. These results indicate that BROWN weathered spoils provide a better growth medium than GRAY unweathered spoils for native trees.

KEYWORDS: mine spoil, forest reclamation approach, mine reclamation, ecological restoration, reforestation

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May 6, 2014

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ACKNOWLEDGMENTS

I would like to thank my parents for making sacrifices to home-school me, providing me with the best education they could. Also, I thank my dear professors from Asbury University, my alma mater—Dr. Stull, Dr. Strait, and Mrs. Witherington taught me so much about life and learning, and I am indebted to them.

I thank Andrea Drayer, Beth Hansen, Mary Weatherford, Millie Hamilton, and Nic Williamson for their help with data collection and analysis. In addition, I am thankful for the help of Kristen McQuerry, who provided much-needed assistance with statistical analyses, and Sarah Hall and Rob Paratley, who helped with species identification. I would also like to thank Patrick Angel, who provided invaluable information, direction, and support to this project. I am also grateful for funding from the Appalachian Research Initiative for Environmental Science (ARIES), which partially sponsored this project.

Finally, I am grateful for the guidance and direction provided by the members of my M.S. thesis steering committee, Dr. Agouridis and Dr. Lhotka, but especially my committee chair, Dr. Barton. He has always been incredibly helpful in pointing me toward resources, providing me with opportunities, and giving me professional feedback.

TABLE OF CONTENTS

Acknowledgments ... iii

List of Tables ... v

List of Figures ... vi

Chapter One: Afforestation success and natural vegetative recolonization ... 1

Introduction ... 1

Methods ... 3

Statistical Methods ... 5

Results and Discussion ... 5

Soil ... 5

Tree Survival ... 11

Tree Growth ... 17

Groundcover ... 20

Conclusions ... 26

Chapter Two: Chemistry and hydrology of interflow ... 27

Introduction ... 27

Methods ... 30

Plot Construction ... 30

Soil ... 31

Trees and Groundcover ... 31

Water Quality ... 32

Hydrology ... 33

Results and Discussion ... 33

Soil ... 33

Trees and Groundcover ... 35

Water Quality ... 39

Hydrology ... 54

Conclusions ... 59

Appendix A: Species composition by spoil type, 2013 ... 61

References ... 64

Vita ... 69

LIST OF TABLES

Table 1.1: Means and standard errors of spoil chemical parameters for 2005 and 2013.............. 6

Table 1.2: Mean tree survival proportions and standard errors for all species by spoil and year 12

Table 1.3: Mean tree survival proportions and standard errors by species and year.................. 14

Table 1.4: Mean survival proportions and standard errors by species, spoil, and year 16

Table 1.5: Tree height means and standard errors for all planted species (green ash, red oak, white oak, and yellow poplar) by spoil type and year 18

Table 1.6: Tree diameter means and standard errors for all planted species (green ash, red oak, white oak, and yellow poplar) by spoil type and year 18

Table 1.7: Tree volume means and standard errors for all planted species (green ash, red oak, white oak, and yellow poplar) by spoil type and year 19

Table 1.8: Mean percent groundcover and total species richness (with standard errors), and calculated Shannon-Weiner index (H') of naturally colonized vegetation by spoil type and year 21

Table 1.9: Distribution of volunteer groundcover vegetation by type (tree/shrub, herb, grass, vine) and native status, expressed as species richness and contribution toward total vegetative cover.. 23

Table 2.1: Means and standard errors of soil pH, electrical conductivity, Mehlich-III-extractable P, K, Ca, Mg, and Zn, cation exchange capacity, texture classes, and exchangeable K, Ca, Mg, and Na.. 34

Table 2.2: Tree volume index (cm^3) and percent survival (with standard errors) by spoil type and year 36

Table 2.3: Percent groundcover and species richness (with standard errors) provided by volunteer vegetation by spoil type and year 38

Table 2.4: Means and standard errors of constituent concentration of water samples by spoil type during the sample period (March 2012 through November 2013).......................... 40

Table 2.5: Coefficients and intercepts (with standard errors) of constituent concentrations in water samples by spoil type (March 2012 through November 2013)............................. 42

Table 2.6: Percentage of rainfall discharged as interflow by plot and season 58

LIST OF FIGURES

Figure 1.1: Soil texture class changes since plot construction 8

Figure 2.1: Electrical conductivity of flow-weighted samples, separated into sampling groups (2005-2007) and (2009-2013) 43

Figure 2.2: Electrical conductivity (EC) measurements are categorized relative to macroinvertebrate toxicity thresholds (Pond et al., 2008)... 45

Figure 2.3: EC of flow-weighted grab samples from BROWN 3 compared to BROWN 1 47

Figure 2.4: Sulfate concentrations in flow-weighted samples, separated into sampling periods 2005-2007 and 2009-2013.. 49

Figure 2.5: Alkalinity (as mg L^{-1} HCO_3^-) of flow-weighted samples, 2009-2013 51

Figure 2.6: pH of flow-weighted samples, separated into sampling periods 2005-2007 and 2009-2013 53

Figure 2.7: Discharged interflow volume summed by season (growing/dormant) and expressed as a percentage of period rainfall 56

CHAPTER ONE

AFFORESTATION SUCCESS AND NATURAL VEGETATIVE RECOLONIZATION

Introduction

Surface mining without appropriate reclamation in Appalachia has been linked to a range of widespread ecological consequences, both terrestrial and aquatic. While recent research has focused on impairment of water resources and aquatic ecosystems (Pond et al. 2008, Smucker and Vis 2011, Hopkins et al. 2013), surface mining also alters terrestrial ecosystems. Wickham et al. (2013) identify a number of "overlooked terrestrial impacts," including direct forest and soil loss, forest fragmentation, and reduced biodiversity. By fragmenting previously intact forest, surface mining can reduce interior forest 1.5-5.0 times as much as it reduces total forest cover (Wickham et al. 2006). Interior forest patches are surrounded by forest, as opposed to other land uses (e.g., open fields), and provide specialized habitat for a number of species (e.g., Cerulean warbler). Also, soil removal during the mining process can suppress natural succession, leading to altered vegetative communities often comprised of nonnative species (Zipper et al. 2011a; Franklin et al. 2012). Because of the effects of surface mining on terrestrial ecosystems, developing techniques capable of restoring native ecosystem structure and function on these disturbed sites is essential.

Reclamation in Appalachia is directed by the Surface Mining Control and Reclamation Act of 1977 (SMCRA). Pre-SMCRA mine reclamation was directed by basic state requirements. Mined sites were often highly unstable and prone to erosion, landslides, and flooding. In light of this, SMCRA emphasized erosion control and land stability on mined sites. Interpretation of these provisions influenced operators to reclaim sites by heavily compacting mine spoils; thus, post-SMCRA soils tend to exhibit high bulk densities (BD) (Conrad et al. 2002, Michels et al. 2007, Zipper et al. 2011a). Also, SMCRA provided provisions that allowed operators to select the best available soil substitute when soil was too thin to stockpile (the predominate case in Appalachia); thus, "mine soil" is often composed of graded overburden with little to no native topsoil (Zipper et al. 2013). As a result, soil on reclaimed mine sites can be chemically and physically unfavorable to native tree growth (Cotton et al. 2012, Zipper et al. 2013).

Because post-SMCRA sites do not support native forests, vegetative cover is typically achieved by seeding nonnative grasses and legumes. These species typically form a competitive groundcover that can further suppress native tree growth. Therefore, reclaimed mined sites tend to remain in arrested succession as nonnative grassland (Zipper et al. 2011b). These grassland patches host novel fauna communities, increase forest edge habitat, increase exotic species invasion risk, and contribute to forest fragmentation and associated interior forest loss (Wickham et al. 2013). In addition to altered ecological structure, ecosystem functions such as carbon sequestration are lower in grassland-reclaimed sites than forest-reclaimed sites (Zipper et al. 2007, Amichev et al. 2008, Shrestha and Lal 2010). Thus, while restoration to native forest is ecologically optimal, reclamation practitioners must overcome a number of barriers with chemically and physically unfavorable soils and vegetative competition.

In response to these issues, a team of scientists and regulatory professionals developed the Forestry Reclamation Approach (FRA): a set of five steps for reclaiming mined sites to encourage native forest regeneration (Zipper et al. 2011b). These steps are: 1) prepare a suitable growth medium, 2) minimize compaction, 3) minimize competition from groundcover, 4) plant early- and late-successional tree species, and 5) use proper tree planting techniques. As previously mentioned, SMCRA permits the use of the best available soil substitute when soil is too thin to stockpile; however, the characteristics of optimal soil substitutes have been poorly defined and have thus been the subject of a body of recent research.

The degree of weathering in topsoil substitutes has been linked to forest establishment and productivity (Emerson et al. 2009, Agouridis et al. 2012, Miller et al. 2012, Zipper et al. 2013). Geologic strata found closer to the surface tend to be more weathered and oxidized, and thus tend to be brown in color. Strata that are buried more deeply in the profile are generally less weathered and more reduced, tending to be grayish. Several ongoing research projects have compared the suitability of weathered and unweathered mine spoils for topsoil substitution in forest reclamation. In a greenhouse study, white ash (*Fraxinus americanus)* and red oak (*Quercus rubra)* demonstrated higher one-season growth on weathered sandstone than unweathered sandstones or shales (Showalter et al. 2010). In a larger field study in West Virginia, average volume of 11 hardwood species was higher in brown sandstone than gray sandstone after three years (Emerson et al. 2009). A follow-up study on this site eight growing seasons after planting confirmed this trend: tree volume in brown sandstone was nearly 10x greater than tree volume in gray sandstone (Wilson-Kokes et al. 2013b). A similar study at a different site in West Virginia also found that tree growth was significantly higher on brown sandstone than gray sandstone after six growing seasons (Wilson-Kokes et al. 2013a). A parallel study was constructed in 2005 on Bent Mountain in Pike County, KY. Similarly, tree volume was significantly higher for 3-year old seedlings in brown sandstone than either gray sandstone or mixed sandstones and shale (Angel 2008).

In addition to improving growth and survival of planted hardwood tree species, reclamation practices that lead to the establishment of native understory vegetation are needed. Competitive nonnative grasses and legumes seeded in conventional reclamation methods tend to inhibit tree growth and survival and can competitively exclude native understory species (Holl 2002, Skousen et al. 2009, Franklin et al. 2012, Lemke et al. 2013). Techniques for the establishment of native understory vegetation on mined sites are not well established. One suggested method for restoration is use of native forest soil, which contains an extensive seed bank of native species (Farmer et al. 1982, Hall et al. 2010). However, as mentioned above, native soil in Appalachia is frequently shallow and therefore not stockpiled during the mining process. Some studies indicate potential for success in seeding a native species mix with tree planting (Franklin and Buckley 2009, Franklin et al. 2012). While other studies in Europe recommend allowing native vegetation to regenerate without human assistance (Hodacova and Prach 2003, Prach and Hobbs 2008, Szarek-Lukaszewska 2009), few studies have addressed unassisted understory recolonization in Appalachia. The objective of this study was to re-assess the influence of spoil type on reforestation success after nine growing seasons (the last assessment being in 2008, following the third growing season), with particular emphasis on identifying continuing trends in tree growth and survival, as well as vegetative understory colonization and species composition.

Methods

Experimental plots were constructed on the Bent Mountain surface mine in Pike County, KY, in 2005 (Angel 2008). Three spoil type treatments (weathered BROWN sandstone, unweathered GRAY sandstone, and MIXED sandstone and shale) were end-dumped in 0.4 ha-plots in duplicate for a total of six plots. End-dumped spoil was graded with one pass of a small bulldozer to strike-off the tops of the spoil piles and create a more level topography (ridge-to-depression depth of 0.5-1.5 m). Excessive grading was avoided to minimize compaction. Four species of hardwood trees were planted directly into these plots: green ash (*Fraxinus pennsylvanica*), red oak (*Quercus rubra*), white oak (*Quercus alba*), and yellow poplar (*Liriodendron tulipifera*). Ground cover seed was not applied to the plots so that natural colonization could be evaluated. Planted trees were monitored by Angel (2008) during the first three growing seasons (2005-2007) and by the current authors in 2013. Tree height was measured using a telescoping measuring rod, and diameter at ground level was measured using calipers. Tree volume index (TVI) was calculated according to the following formula: TVI = d^2h, where d = diameter and h = height. The number of surviving planted trees was recorded for each plot.

Soil samples were collected by Angel (2008) in the first three years (2005-2007) and by the current authors in January 2013. Each plot (n = 6) was subdivided into 16 subplots, eight of which were randomly selected as subsampling locations. In each subplot, six samples were collected to a depth of 6-8 cm using a sampling trowel and thoroughly mixed on site to give a composite subsample representative of both ridge and depression areas. Thus, eight subsamples were collected per plot, for a total of 48 samples. 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). P, K, Ca, Mg, and Zn were analyzed by Mehlich-III extraction (Soil and Plant Analysis Council, 2000). Total N was analyzed with LECO combustion. Soil EC was determined in a soil water extract (Soil and Plant Analysis Council, 2000). Exchangeable bases (K, Ca, Mg, and Na) and cation exchange capacity were determined by the ammonium acetate method (Summer and Miller 1996).

Understory vegetation was characterized by Angel from 2005-2007 and by the current authors in August 2013 according to the method of Farmer et al. (1981). In this method, the observer tallies groundcover vegetation “hits” on a scored milacre radius (describes a circle equal to a thousandth of an acre) stick at regular intervals from a random plot entry point. A point was scored as a “hit” if a raindrop falling on the point would be intercepted by vegetation or litter before reaching bare rock or soil. This vegetative interception could be performed by canopy trees, midstory trees/shrubs, or understory vegetation or litter. These data were used to calculate species richness, invasive/native species richness, and species composition by groundcover. Individual plants were identified following the nomenclature outlined in Jones (2005). Additional species noted by the observer but not present in any of the sampling plots were recorded as present and incorporated in the species richness metric, but were not assigned a groundcover value. Species richness and diversity (Shannon–Weiner index, H’) were calculated. The Shannon-Weiner index is normally calculated using the number of individuals of a given species relative to total number of individuals surveyed. However, because our survey only determined groundcover percentage, we calculated H' using groundcover of a given species relative to total vegetative cover.

Statistical methods:

Means were calculated by plot for each soil analysis parameter, and these were analyzed using PROC MIXED (SAS 9.3), with spoil type (BROWN, GRAY, MIXED) and year (2005, 2006, 2007, 2013) as fixed effects and replicate as random effect. Mean tree height, diameter, and volume were calculated for each plot, and these were also analyzed using PROC MIXED (SAS 9.3), with spoil type, year, and species as fixed effects and replicate as random effect. Mean tree survival proportions were calculated for each plot, and these was analyzed using PROC GLIMMIX (SAS 9.3), with spoil type and tree species (green ash, red oak, white oak, yellow poplar) as fixed effects and replicate as random effect.

Results and Discussion:

Soil:

Soil texture is an important determinant of soil physical and chemical properties, especially on highly disturbed sites undergoing early pedogenesis. Particularly, development of clay particles is of interest. Clay is the smallest soil particle size class and, thus, has the highest surface area to mass ratio. The high surface area of clay provides more space for interaction with water and soil chemicals, particularly base cations. Thus, soils with high clay fractions are typically characterized by high water holding capacity and high cation exchange capacity (CEC), both of which are important for soil-plant interactions and are desirable markers of soil development during pedogenesis. While the clay fraction decreased over the first three years post-construction in BROWN (Angel, 2008), it appears to have rebounded and stabilized at slightly above 10% (Table 1.1).

Table 1.1: Means and standard errors of spoil chemical parameters for 2005 and 2013, by spoil type. Means with the same letter are not significantly different within the given year.

	2005			2013		
Parameter	Brown	Gray	Mixed	Brown	Gray	Mixed
pH	6.03b ± 0.47	8.07a ± 0.13	8.33a ± 0.01	6.07b ± 0.46	8.44a ± 0.29	8.12a ± 0.08
EC ($\mu S\ cm^{-1}$)	163a ± 21	160a ± 1.6	185a ± 6.5	73a ± 8.1	78 a ± 21	87a ± 0.6
P ($mg\ kg^{-1}$)	6.09a ± 0.09	2.62b ± 1.0	0.53b ± 0.03	5.47a ± 0.84	2.47b ± 1.47	1.47 b ± 0.03
K ($mg\ kg^{-1}$)	51.1a ± 3.66	50.0a ± 5.78	52.5a ± 6.06	62.6a ± 1.03	27.6b ± 1.81	56.3a ± 5.22
Ca ($mg\ kg^{-1}$)	573b ± 133	1080ab ± 292	1590a ± 28	653a ± 97	838a ± 465	887a ± 94
Mg ($mg\ kg^{-1}$)	368a ± 8.78	295a ± 75.5	251a ± 10.5	304a ± 3.94	282a ± 104	274a ± 14.1
Zn ($mg\ kg^{-1}$)	2.33a ± 0.08	3.77a ± 0.73	3.05a ± 1.5	2.78a ± 0.38	4.68a ± 0.9	6.34a ± 1.21
CEC ($meq\ 100g^{-1}$)	8.27a ± 1.72	2.54b ± 0.38	3.38b ± 0.42	7.16a ± 1.29	3.27b ± 0.23	4.69b ± 0.50
Exch K ($meq\ 100g^{-1}$)	0.07a ± 0.00	0.06a ± 0.00	0.08a ± 0.01	0.14a ± 0.00	0.05c ± 0.00	0.10b ± 0.02
Exch Ca ($meq\ 100g^{-1}$)	1.46a ± 0.61	2.18a ± 1.2	2.89a ± 0.23	2.61a ± 0.47	1.84a ± 0.65	2.99a ± 0.92
Exch Mg ($meq\ 100g^{-1}$)	2.14a ± 0.04	1.23b ± 0.61	0.89b ± 0.18	2.04a ± 0.03	0.87b ± 0.10	1.32b ± 0.24
Exch Na ($meq\ 100g^{-1}$)	0.15a ± 0.04	0.07a ± 0.01	0.06a ± 0.00	0.23a ± 0.00	0.19a ± 0.05	0.19a ± 0.05
Total N (%)	-	-	-	0.09a ± 0.02	0.03a ± 0.00	0.03a ± 0.00
% Sand	60.8b ± 0.63	77.8a ± 0.63	73.9b ± 1.26	64.1b ± 1.69	74.2a ± 0.73	57.4b ± 3.68
% Silt	27.2a ± 0.66	15.7b ± 0.35	18.4a ± 0.32	25.6a ± 0.61	18.8b ± 0.43	33.4a ± 3.26
% Clay	11.9a ± 2.84	6.5b ± 1.76	7.7b ± 1.09	10.3a ± 1.08	7.0b ± 0.30	9.1b ± 0.42

Wilson-Kokes et al. (2013b) also reported higher percent fines in brown spoil than gray spoil after eight growing seasons in their study in West Virginia. Consistent with these observations, both CEC and field capacity were higher in BROWN than either GRAY or MIXED. Higher CEC in BROWN (7.18 meq $100g^{-1}$) compared to GRAY (3.27 meq $100g^{-1}$) and MIXED (4.69 meq $100g^{-1}$) is a factor of greater clay fraction and indicates that BROWN is able to hold more cations (Table 1). This relationship is reflected in observations of individual elemental concentrations, discussed below. Also, field capacity is higher ($p < 0.05$) in BROWN (19.3%) than either MIXED (14.0%) or GRAY (11.5%). This relationship indicates that BROWN is capable of higher water retention than either GRAY or MIXED, which is important on these highly disturbed sites given their xeric location (e.g. open conditions with direct sunlight and low moisture). Angel (2008) observed lower soil moisture on GRAY and MIXED than BROWN, and attributed this to their lower field capacity and clay content. The other soil texture trend of note is the marked increase in silt content in MIXED from 2005 to 2013 (Figure 1.1).

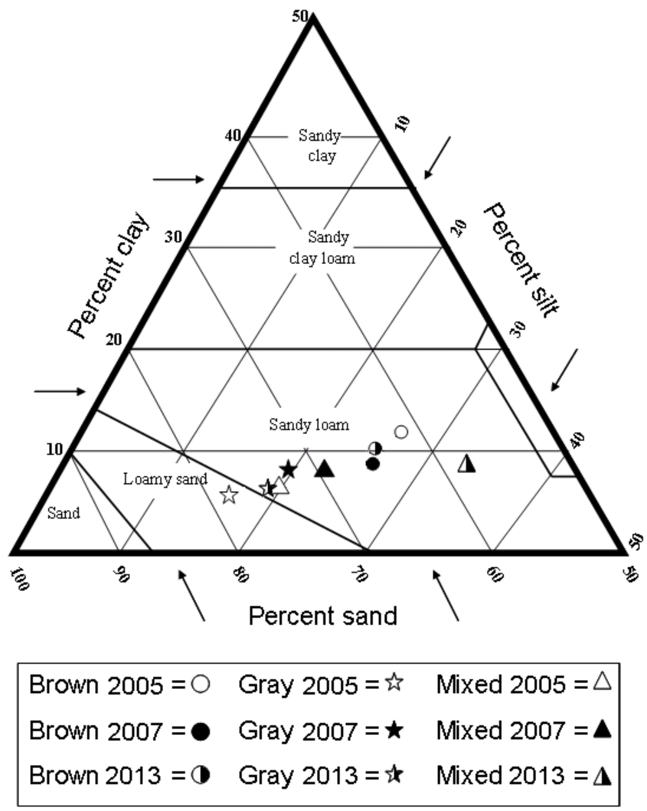

Figure 1.1: Soil texture class changes since plot construction. While BROWN and GRAY have remained relatively stable texturally, changes in soil texture in MIXED evidence weathering of shales, contributing to a rapid increase in silts.

Miller et al. (2012) also found that development of soil fines was much more rapid in mine spoils containing shale (e.g., shale treatment and mixed shales/sandstones) than in either brown sandstone or gray sandstone. They reported high silt and clay fractions (38.1% silt and 14.9% clay) in shale spoil, compared to intermediate levels in the mixed sandstone and shales (21% silt and 7.8% clay) and low levels in brown and gray sandstones (< 17% silt and <5% clay). They predicted that soil particle size distribution may mimic natural conditions in shale spoils more quickly than in sandstone. These field observations were supported by laboratory slake-durability tests, which also demonstrated that shale breakdown was more rapid than either brown or gray sandstone. Consistent with Miller et al. (2012), we observed higher silt fraction in MIXED than either BROWN or GRAY (Table 1.1), and we predict that this trend will be followed by increases in CEC and water holding capacity over time. These changes in soil particle size distribution are an important indication that soil forming processes are actively occurring on these sites.

One of the most important soil chemical trends observed was a decrease in electrical conductivity (EC) concentration (Table 1.1). Soil EC has decreased from >150 $\mu S\ cm^{-1}$ in all treatments in 2005 to <90 $\mu S\ cm^{-1}$ in 2013. This trend in soil EC was mirrored by decreasing EC in water draining from the plots (See Chapter 2) and suggests that the pool of highly reactable/dissolvable salts has decreased drastically since plot construction. Wilson-Kokes et al. (2013b) also reported reduced soil EC (30-60 $\mu S\ cm^{-1}$) in brown and gray sandstones after eight growing seasons. Zipper et al. (2013) identify soil EC as an important determinant of tree growth on reclaimed sites. In general, soils with high EC are less favorable to plant growth than more dilute soils. Although weathered spoils tend to have lower EC than unweathered spoils, we observed that EC in BROWN was not significantly lower than GRAY or MIXED (Table 1.1). Agouridis et al. (2012) proposed that pyritic materials were responsible for high EC in interflow draining from BROWN. The presence of pyritic materials in BROWN would also explain why EC in BROWN is not lower than EC in less weathered spoils, as is typical in other studies (Roberts, 1988; Showalter et al. 2010). In all treatments on Bent Mountain, soil EC was not high enough (~400 $\mu S\ cm^{-1}$) to cause salt stress (Rendig and Taylor 1989), and was within the range of native eastern Kentucky soils found in yellow-poplar and white oak stands (Cotton 2006).

Angel (2008) observed that soil pH increased slightly in all spoil types over the first three years. Shrestha and Lal (2007) concluded that high pH in minesoils was related to the presence of unweathered carbonates. Carbonate weathering, in addition to producing free bases (e.g., Ca, Mg) which can be lost to leaching, also produces bicarbonate ions (HCO_3^-), which raise pH. Maharaj et al. (2007) attributed declining cation concentrations (Ca, Mg) to carbonate weathering; continued weathering is expected to deplete available unweathered carbonates, thus allowing pH to decline. Ward (2009) also found that calcite weathering was occurring more rapidly on BROWN than GRAY or MIXED on Bent Mountain. More rapid weathering and subsequent leaching of carbonates in BROWN is consistent with our observation of lower spoil pH in BROWN (6.1), than GRAY (8.4) and MIXED (8.1) (Table 1.1). Maharaj et al. (2007) observed that the rate of weathering would increase with tree root penetration (opening new flowpaths) and subsequent respiration (producing carbonic acid). These biological weathering processes would be accelerated on BROWN compared to GRAY or MIXED, given greater tree growth on that spoil type.

Other important chemical constituents, P and K, tend to be higher in BROWN than GRAY or MIXED. Wilson-Kokes et al. (2013b) attributed low P in unweathered spoils to formation of insoluble Fe complexes and low K to leaching, both of which are potential explanations for low P and K levels in GRAY and MIXED (Table 1). Calcium (Ca) levels are high in GRAY and MIXED relative to BROWN; however, Ca is increasing in BROWN while decreasing in both GRAY and MIXED. CEC may play a role in rising Ca and exchangeable base concentrations in BROWN. As weathering progresses, as suggested by lowering EC concentrations, leaching and translocation of the mineral constituents increases (Maharaj et al. 2007). Since BROWN exhibits a higher CEC than GRAY or MIXED (Table 1), it may have a higher affinity to retain these mobile cations.

Additional samplings will be required to characterize long-term pH trends on MIXED. It is possible, given observed declines in soil EC, that the pool of soluble carbonates and other readily available minerals is shrinking in MIXED. If carbonate weathering is slowing in MIXED, it is possible that pH will continue to drop due to a decrease in soil buffering capacity. Soil pH in GRAY decreased from 2007 to 2013 (data not shown), as was observed in BROWN and MIXED. Further monitoring will be necessary to evaluate if and when carbonate weathering rates stabilize in GRAY. With greater biological inputs (e.g., respiration giving CO_2, which reacts with water to give carbonic acid), on BROWN, we project that soil will continue to acidify. Additional surveys will be required to identify if and when pH on BROWN reaches the pH range (4.89 – 5.65) of native Appalachian soil (Cotton et al. 2012).

Tree Survival:

When overall survival (for all species) is considered by spoil type, Angel observed that 2006 survival was higher in GRAY (96%) than MIXED (85%) or BROWN (82%) (Table 1.2).

Table 1.2: Mean tree survival proportions and standard errors for all species by spoil and year. Mean survival proportions with the same letter are not significantly different (p = 0.05) among spoil types in the same year.

Survival	2006	2007	2013
Brown	0.822b ± 0.01	0.870a ± 0.00	0.864a ± 0.02
Gray	0.958a ± 0.01	0.878a ± 0.03	0.644a ± 0.15
Mixed	0.854b ± 0.01	0.813a ± 0.02	0.679a ± 0.05

By 2013, although survival difference among spoil types was not significant, we observed a distinct decreasing trend in survival on both GRAY and MIXED, while survival remained relatively constant in BROWN (Table 1.2). While initial survival on GRAY and MIXED was high, conditions on these sites were harsh. Angel (2008) reported that soil pH was greater than 8.0, which is above the recommended range for all four planted species (USDA 1973, Williston and LaFayette 1978). In addition, Angel (2008) observed that soil moisture in GRAY and MIXED was low, and he observed evidence of surface sealing in GRAY. These conditions are unfavorable to the growth of native hardwoods. It is likely that these stresses have cumulatively led to the observed decline in survival on GRAY and MIXED. Additional monitoring will be required to see if survival stabilizes or continues to decline on these treatments.

Tree survival comparisons by species through nine growing seasons are reported in Table 1.3.

Table 1.3: Mean tree survival proportions and standard errors by species and year. Means with the same letter are not significantly different (p = 0.05) within species.

Survival	2006	2007	2013
Green Ash	0.950a ± 0.02	0.966a ± 0.01	0.945a ± 0.02
Red Oak	0.895a ± 0.04	0.766b ± 0.01	0.728b ± 0.03
White Oak	0.922a ± 0.05	0.996a ± 0.06	0.654b ± 0.01
Yellow-Poplar	0.745b ± 0.04	0.686b ± 0.03	0.588b ± 0.07

Survival of green ash has been highest since 2006, indicating that the tree is an excellent candidate for survival on reclamation sites in the region. Other studies have noted high survival for ash on mined land through the first few growing seasons (Angel et al. 2006, Skousen et al. 2007, Emerson et al. 2009, Miller et al. 2012). This being noted, the continued expansion of the emerald ash borer (EAB) poses a serious threat to this and other native ash species in the eastern U.S (Kashian and Witter 2011, Flower et al. 2013). Continued inclusion of ash in reclamation plantings may be advisable, especially in the event that EAB does not persist, or that EAB treatment options become more commonplace and effective. Also, because of its high survival on these sites, research is necessary to identify whether ash provides ecological benefits as a “founder” species. Red oak survival appears to have stabilized around 73%, which is slightly higher than the 60% survival reported by Wilson-Kokes et al. (2013b). Yellow poplar survival (58.8%) was also equivalent to Wilson-Kokes et al. (2013b) at 52%.

White oak survival precipitously declined (>30%) from 2007 to 2013 (Table 1.3). While PROC GLIMMIX found no significant interaction between spoil and species for 2013 ($p > 0.05$), note that white oak survival in GRAY (48%) and MIXED (46%) was half as high as white oak survival in BROWN (>95%) (Table 1.4).

Table 1.4: Mean survival proportions and standard errors by species, spoil, and year. Increase in survival from year to year indicates that trees were misclassified as dead in an earlier survey. GA = green ash; RO = red oak; WO = white oak; and YP = yellow poplar.

	GA	RO	WO	YP
		BROWN		
2006	0.929 ± 0.02	0.859 ± 0.06	0.807 ± 0.00	0.699 ± 0.06
2007	0.969 ± 0.00	0.771 ± 0.02	1.0 ± 0.11	0.699 ± 0.09
2013	0.941 ± 0.01	0.781 ± 0.02	1.0 ± 0.16	0.653 ± 0.08
		GRAY		
2006	0.961 ± 0.00	1.0 ± 0.00	1.0 ± 0.06	0.852 ± 0.00
2007	0.961 ± 0.00	0.799 ± 0.01	1.0 ± 0.12	0.725 ± 0.06
2013	0.906 ± 0.02	0.743 ± 0.09	0.469 ± 0.22	0.537 ± 0.22
		MIXED		
2006	0.952 ± 0.05	0.810 ± 0.01	0.975 ± 0.09	0.688 ± 0.00
2007	0.956 ± 0.04	0.729 ± 0.02	0.926 ± 0.12	0.654 ± 0.00
2013	0.970 ± 0.02	0.669 ± 0.04	0.480 ± 0.11	0.623 ± 0.05

While white oak persisted relatively well through the first three years after planting, survival dropped off in subsequent growing seasons. White oak prefers acidic soil conditions (4.5-6.2), and its highest recommended limit is 7.7 (USDA 1973, (Williston and LaFayette 1978). Therefore, it is likely that the stressful chemical (e.g., high pH, low nutrient availability) and physical (e.g., low moisture availability) conditions on GRAY and MIXED exerted a cumulative effect on this species, resulting in the significant decline. These plots should be monitored into the future to see if a similar decline is observed in the other species.

Tree Growth:

Height and diameter at base were measured for all planted species on each plot and means for spoil type are reported in Tables 1.5 and 1.6. Tree volume indices are reported in Table 1.7. In 2005, tree height, diameter, and volume were similar across spoil types. By 2007, growth was significantly higher in BROWN than either GRAY or MIXED (Angel, 2008). In 2013, growth on BROWN was higher than either GRAY or MIXED. We also observed that growth on MIXED had outpaced growth on GRAY for all species but white oak. This suggests that MIXED is a candidate for an intermediate-quality soil substitute between low-quality gray spoils and higher quality brown spoils.

Table 1.5: Tree height means and standard errors for all planted species (green ash, red oak, white oak, and yellow poplar) by spoil and year. Means with same letter are not statistically different (p = 0.05) among spoil types within year.

Height (cm)	2005			2013		
Species	Brown	Gray	Mixed	Brown	Gray	Mixed
Green Ash	30.5a ± 1.0	32.0a ± 0.4	32.2a ± 0.1	303a ± 12.9	65.9c ± 14.8	143b ± 7.4
Red Oak	35.2a ± 1.4	36.8a ± 0.9	36.1a ± 0.2	293a ± 10.5	42.7c ± 17.5	106b ± 6.5
White Oak	30.0a ± 1.3	27.9a ± 0.0	30.0a ± 0.3	268a ± 11.4	23.9b ± 10.9	42.6b ± 8.5
Yellow Poplar	50.5a ± 0.7	49.3a ± 0.4	49.1a ± 2.6	410a ± 7.8	57.9c ± 9.9	127b ± 0.4
All	36.8a ± 0.6	36.3a ± 0.7	37.0a ± 0.3	314a ± 7.8	50.5c ± 13.6	114b ± 3.3

Table 1.6: Tree diameter means and standard errors for all planted species (green ash, red oak, white oak, and yellow poplar) by spoil type and year. Means with the same letter are not statistically different (p = 0.05) among spoil types within year.

Diameter (mm)	2005			2013		
Species	Brown	Gray	Mixed	Brown	Gray	Mixed
Green Ash	5.4a ± 0.2	5.7a ± 0.1	5.6a ± 0.0	41.8a ± 1.2	11.8c ± 2.2	21.1b ± 1.6
Red Oak	5.7a ± 0.1	5.6a ± 0.0	5.4a ± 0.2	48.0a ± 0.0	11.1c ± 3.3	23.2b ± 1.0
White Oak	4.8a ± 0.2	4.4a ± 0.2	4.7a ± 0.1	44.9a ± 5.5	6.7b ± 2.6	9.9b ± 2.1
Yellow Poplar	6.6a ± 0.2	7.0a ± 0.1	6.4a ± 0.2	64.5a ± 3.0	15.1c ± 2.3	29.5b ± 2.6
All	5.7a ± 0.1	5.7a ± 0.1	5.6a ± 0.0	48.7a ± 0.6	11.3c ± 2.8	22.0b ± 1.3

Table 1.7: Tree volume means and standard errors for all planted species (green ash, red oak, white oak, and yellow poplar) by spoil type and year. Means with the same letter are not statistically different (p = 0.05) among spoil types within year.

Volume (cm^3)	2005			2013		
Species	Brown	Gray	Mixed	Brown	Gray	Mixed
Green Ash	11.6a ± 1.9	14.6a ± 0.2	13.4a ± 0.6	8,006a ± 890	302c ± 105	1,420b ± 11
Red Oak	13.4a ± 1.0	14.0a ± 0.7	13.0a ± 1.0	10,524a ± 990	162c ± 127	1,399b ± 87
White Oak	8.4a ± 1.4	6.8a ± 0.2	7.8a ± 0.2	8,982a ± 1890	68b ± 56	134b ± 72
Yellow Poplar	29.7a ± 0.8	30.0a ± 1.0	26.3a ± 3.1	25,241a ± 2410	380c ± 147	4,183b ± 1280
All	16.1a ± 1.1	16.2a ± 1.0	15.2a ± 0.4	12,270a ± 292	237c ± 115	1,837b ± 277

Wilson-Kokes et al. (2013b) found that tree growth was 10-12 times better on uncompacted brown spoil (3,913-5,182 cm^3) than gray spoil (407 cm^3) through eight growing seasons. Our results are even more extreme than these, with average tree volume on BROWN (12,270 cm^3) fifty times greater than tree volume on GRAY (237 cm^3) after nine years.

Of the 11 species planted by Wilson-Kokes et al. (2013b), three were also planted on Bent Mountain: white oak, red oak, and yellow poplar. In their study, in uncompacted brown spoil treatments, white oak tended to have the greatest volume out of the three species (3,027-5,392 cm^3), followed by red oak (3,609-3,181 cm^3) and yellow poplar (1,682-3,910 cm^3). Interestingly, this trend is completely reversed on Bent Mountain, with yellow poplar experiencing the greatest growth (25,241 cm^3) and white oak the lowest growth (8982 cm^3).

Groundcover:

Angel (2008) observed by summer 2006 that plant colonization of the Bent Mountain plots was occurring by natural means (Table 1.8).

Table 1.8: Mean percent groundcover and total species richness (with standard errors), and calculated Shannon-Weiner index (H') of naturally colonized vegetation by spoil type and year. Values with the same letter are not statistically different (p = 0.05) among spoil types within year.

	Spoil	Groundcover (%)	Species Richness	H'
2006	Brown	42.3a ± 0.03	40a ± 1	1.64
	Gray	1b ± 0.00	6b ± 0.5	-
	Mixed	2.6b ± 0.00	21b ± 2	0.94
2007	Brown	66.4a ± 0.06	61a ± 8	2.59
	Gray	2b ± 0.00	12c ± 0.5	1.53
	Mixed	5.8b ± 0.00	35b ± 0.5	2.58
2008	Brown	78.6a ± 0.00	70a ± 10	2.68
	Gray	3.5b ± 0.00	18c ± 1.5	1.83
	Mixed	8.6b ± 0.00	41b ± 1	2.77
2013	Brown	99.1a ± 0.00	57a ± 5	2.34
	Gray	9.8c ± 0.05	42b ± 4.5	2.10
	Mixed	20.2b ± 0.07	43b ± 0.5	1.84

In 2006, naturally colonizing vegetation on BROWN provided 42.3% groundcover, which was greater than groundcover on either GRAY (1%) or MIXED (2.6%). By 2008, groundcover on BROWN had reached 78.6%; however, this rapid increase in vegetative cover was not mirrored in GRAY (3.5%) or MIXED (8.6%). In 2013, cover on BROWN was nearly 100%, ten times greater than GRAY (9.8%) and nearly five times greater than MIXED (20.2%). Wilson-Kokes et al. (2013b) observed similar trends, with 70% cover on BROWN and 10% cover on GRAY. Interestingly, the study by Wilson-Kokes et al. (2013b) seeded plots two years after planting with a tree-friendly mix of redtop (*Agrostis gigantean*), birdsfoot clover (*Lotus corniculatus*), and perennial ryegrass (*Lolium perenne*). While some studies suggest that seeding a tree-friendly groundcover mix may increase tree survival and growth (Franklin and Buckley, 2009), other studies have found that introducing groundcover species during reclamation may inhibit tree growth (Skousen et al., 2009). Our findings would suggest that seeding may be unnecessary if soil erosion is not anticipated to be problematic, such as in flat loose-dumped spoil (Miller et al., 2012). Additional research is necessary to elucidate the interaction of planted groundcover species with planted trees.

In addition to higher percent groundcover, the naturally regenerated vegetative community on BROWN had significantly higher species richness than GRAY or MIXED in all surveyed years. Across treatments, native species richness ranged from 68% (BROWN) to 55% (GRAY) of total species richness. Native species provided only 35.2% of total vegetative cover in BROWN, compared to 96.6% in GRAY and 76.1% in MIXED (Table 1.9).

Table 1.9: Distribution of volunteer groundcover vegetation by type (tree/shrub, herb, grass, vine) and native status, expressed as species richness and contribution toward total vegetative cover.

	BROWN	GRAY	MIXED
Grass Species	5	3	3
% Cover by Grasses	6.2%	67.4%	61.9%
Herb Species	20	17	16
% Cover by Herbs	52.1%	33.8%	26.1%
Tree/Shrub Species	27	17	19
% Cover by Trees/Shrubs	47.6%	8.8%	17.8%
Vine Species	5	5	5
% Cover by Vines	0.9%	4.7%	1.4%
Native Species	39	26	29
% Cover by Natives	35.2%	96.6%	76.4%
Total Species	**57**	**42**	**43**

Cover in BROWN was dominated by Chinese lespedeza (*Lespedeza cuneata*) (42.8%), a highly competitive nonnative species common to disturbed areas. The next greatest contributor to cover was black locust (*Robinia pseudoacacia*) (13.3%), followed by royal paulownia (*Paulownia tomentosa*) (9.8%) and autumn olive (*Eleagnus umbellata*) (8%). In contrast, GRAY and MIXED are dominated by a native broomsedge (*Andropogon virginicus*), which provides 60% of total vegetative cover on both spoils. Broomsedge is commonly found in open areas such as abandoned fields, overgrazed pastures, cut-over timber sites, and rights of way and it grows on a wide variety of soils, preferring loose, sandy, moist sites with low fertility. On infertile soils, broomsedge is a long-lived competitor that has allelopathic chemicals which can have adverse affects on other plants trying to colonize (Rice, 1972).

While it is disheartening to see that nonnative species are dominating the vegetative cover of BROWN, we were encouraged to observe that native volunteer species (e.g., black locust) and planted species (green ash, red oak, white oak, and yellow poplar) were not suppressed by heavy lespedeza competition or canopy exclusion by paulownia and autumn olive. We observed several locations within BROWN plots where planted species had attained complete canopy closure. In these areas, the understory was completely transformed compared to unshaded areas. Dead matter from lespedeza was evident; however, little to no lespedeza was thriving in these locations. We conclude that, although it was originally present in the area, shading by tree species caused lespedeza to decline. In addition, we observed that several shade-tolerant species were thriving under canopy closure, including wild hydrangea (*Hydrangea sp.*), alum-root (*Heuchera sp.*) and Christmas fern (*Polystichum acrostichoides*). Also, we observed several instances where planted tree species were growing through and above nonnative trees and shrubs.

It is difficult to predict at this point how the observed invasive species will affect long-term forest development on these sites. Yates et al. (2004) found that autumn olive is able to colonize and persist in interior forest; thus, this species may persist on BROWN even if total canopy cover is achieved. In contrast, Lemke et al. (2013) found that Chinese lespedeza tends to be found only in open, sunny areas, consistent with our observations that the species appears to be excluded from areas undergoing canopy closure. Finally, Longbrake and McCarthy (2001) found that royal paulownia prefers sunny sites over shade. Eventual canopy closure may inhibit persistence of this species over time. Follow-up studies on the vegetative development of these sites are necessary to project how invasive species will interact with the developing ecosystem.

If these species suppress establishment of native forest conditions, management of invasive species on naturally regenerating reclaimed mine sites may be necessary to ensure restoration to a native ecosystem.

Also of note is the difference in species and cover distribution among species types (trees/shrubs, grasses, herbs, and vines). As noted above, three fifths of total vegetative cover in GRAY (9.8%) and MIXED (20.2%) is provided by *A. virginicus*. It is not surprising, then, that three grass species provide 61.9% of cover on MIXED and 67.4% of cover on GRAY. In contrast, five grass species provide only 6.2% of vegetative cover on BROWN. Cover on BROWN, while largely composed of herbs (52.1%) is also heavily influenced by tree and shrub species (47.6%). This is in stark contrast to GRAY (8.8%) and MIXED (17.8%). Future vegetative analyses on BROWN should employ more sophisticated techniques than Farmer et al. (1981) to adequately incorporate multi-story composition.

When 2013 species composition on BROWN is compared to surveys conducted by Angel (2008) in 2006, 2007, and 2008 (data not shown), several trends become apparent. First, several sun-tolerant species that were identified on BROWN in previous years were not found on BROWN in 2013. These included several species of grasses and disturbance-specialist herbaceous species such as *Phytolacca americana* (pokeweed), *Tussilago farfara* (coltsfoot), and *Chenopodium album* (lambsquarters). In addition to observed declines in sun-tolerant species, we observed an increase in more shade-tolerant herbaceous species, such as *Heuchera sp*. (alum root) and *Polystichum acrostichoides* (Christmas fern). Also, midstory and understory trees and shrubs, including *Cornus drumondii* (roughleaf dogwood), *Prunus serotina* (black cherry), *Symphoricarpus orbiculatus* (coralberry), and *Hydrangea sp*. (wild hydrangea) were observed to be increasing in groundcover and species richness on BROWN. The decline in sun-tolerant species and concomitant increase in shade-tolerant species suggests that shading by lespedeza and, more importantly, by developing forest canopy is altering species composition. These observations indicate that succession to mesic forest is well underway on BROWN plots.

Kentucky law requires that adequate groundcover (a minimum of 70%) be present on mines reclaimed for non-commercial forest post-mining uses (Eddins 1983). Necessary groundcover is typically attained by intentionally seeding with disturbance-tolerant species. However, the potential for natural vegetative regeneration has not been well-researched in the Appalachian region. Angel (2008) found that BROWN met state groundcover requirements for non-commercial forest post-mining land use after only three growing seasons. In contrast, both

GRAY and MIXED did not meet minimum groundcover requirements after nine growing seasons. These results indicate that natural colonization may provide sufficient groundcover on high-quality soil substitutes (e.g., brown weathered sandstone) in areas with low potential for erosion. In contrast, lower quality soil substitutes (e.g., gray unweathered sandstone) may require intentional seeding of hardy groundcover species to encourage attainment of groundcover standards.

Conclusions

Tree growth and groundcover were both greater in BROWN than GRAY or MIXED. Observed plant preference for weathered brown spoil is likely related to more soil-like chemical and physical growing conditions provided by BROWN in contrast to chemically and physically unfavorable conditions on GRAY and MIXED. These observations confirm the conclusions made by Angel (2008) in the early phases of development on these plots, and by other researchers on other similar studies (e.g., Wilson-Kokes et al. 2013b, Zipper et al. 2013) that weathered brown spoils provide a better growing medium than unweathered gray spoils in Appalachia. These data provide direction for a critical current issue in Appalachia: restoration of native ecosystem structure and function on surface mined land. Restoration of native hardwoods requires the development of favorable soil on disturbed sites, and this study indicates that soils that are chemically and physically favorable for native tree growth does not develop from unweathered spoils through eight growing seasons. These observations should contribute to adopting improved mining and reclamation policy directing industry to preferentially utilize weathered brown spoils when reclaiming mined land in Appalachia. These authors would also recommend continued monitoring of the Bent Mountain site over time to ascertain whether tree growth on BROWN will continue to outpace growth rates on GRAY and MIXED, whether tree persistence will drop off on GRAY and MIXED relative to BROWN, and how soil will continue to develop on MIXED and BROWN (especially with regard to percent fines). Such studies will provide critical information on mine spoil influence on long-term development of reclaimed mined land.

CHAPTER TWO

CHEMISTRY AND HYDROLOGY OF INTERFLOW

Introduction

Surface mining for coal has been tied to a broad spectrum of environmental impairments in the Appalachian region. 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 minesoils 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 hydrologic function 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 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. Thus, the heavy compaction associated with typical surface mine reclamation produced a flashier hydrograph. Another study in Maryland (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/hr, compared to 30 cm/hr 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). Because of 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 contribute to raising 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, Se) (Lindberg et al. 2011). Also, researchers continue to implicate mine drainage in declines of sensitive macroinvertebrate taxa in eastern Kentucky headwater streams (Pond 2010, 2011) and altered biofilm function/development in southeast Ohio (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 impaired hydrologic function 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 (Burger et al., 2005; 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, Wilson-Kokes et al. 2013b). 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, 1032, and 1224, 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 hydrologic function. On the Bent Mountain study site, Taylor et al. (2009) 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. (2009) reported that there was no difference among the hydrologic function of the different spoil types (BROWN, GRAY, 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). By 2013, average planted tree volume on BROWN had increased from 235 cm^3 (2007) to 12,270 cm^3 while tree volume on MIXED and GRAY were much lower (1,837 cm^3 and 237 cm^3, respectively) (See Chapter 1). With this drastic increase in tree volume, we predict that evapotranspiration will likely play an increased role in water budgeting, especially on BROWN plots.

This study was designed to follow-up on the previous work by Agouridis et al. (2012), Angel (2008), and Taylor (2009a,b). Similar studies (e.g., Wilson-Kokes et al., 2013) 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.

Methods

Plot construction:

Experimental plots were constructed on the Bent Mountain surface mine in Pikeville, KY, in 2005 (Angel, 2008). Three spoil type treatments (brown weathered sandstone, gray unweathered sandstone, 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 EC logger) and was drained into an underlying deep mine to effectively isolate the plots from each other. 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 americanus*), 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).

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, 8 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). P, K, Ca, Mg, and Zn were analyzed by Mehlich-III extraction (Soil and Plant Analysis Council, Chapter 7, 2000). Soil EC was determined in a soil water extract (Soil and Plant Analysis Council, Chapter 5, 2000). Cation exchange capacity was determined by the ammonium acetate method (Summer and Miller 1996).

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

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.

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 UK Department of Forestry Hydrology Lab for analysis. Water pH and alkalinity were measured using an auto-titrater. 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).

In addition, a Hobo® conductivity logger (Onset®) was installed at each data acquisition station, and recorded EC every 30 minutes. These data tended to be noisy, with long periods of low-range EC values (e.g., < 50 μS/cm) punctuated by periods of higher flow (e.g., > 100 μS/cm). In the lab, we found that a logger recorded values for EC (ranging as high as 20 μS/cm) when placed in a moist environment, but with the sensor not in direct contact with water. We concluded that condensation could be responsible for false EC values in our field experiment. To correct for these, we established a threshold of 50 μS/cm below which data points would be excluded from our analysis. We based this threshold on our observations of our data, which showed that clearer signals tended to be above 50 μs/cm, and on published values for regional reference-quality streams (e.g., Pond et al. 2008).

After eliminating datapoints flagged as false, a daily average was computed for every day with 24 or more valid observations (suggesting that water was flowing at least half of the day). Average daily EC were analyzed by PROC MIXED, with treatment and day (since first sample in current period) and treatment*day interaction modeled as fixed effects, and rep (n = 2) modeled as a random effect. The spatial power SP(POW) covariance structure provided the best fit for uneven repeated measures sampling intervals. Average daily EC was compared to the 300-500 $\mu S\ cm^{-1}$ threshold observed by Pond et al. (2008).

Hydrology:

Tipping bucket data were downloaded during regular sampling trips. Due to occasional equipment failure (e.g., data loggers reaching maximum capacity, and tipping bucket malfunctions), accurate data were not obtained for all plots for the entire study period. In light of this, only periods during which accurate data were available for all plots were analyzed during this study. Flow data were separated into growing and dormant seasons according to published growing season dates for Pike County, KY: April 20-October 26 (NRCS Soil Survey for Pike Co., KY). Total flow (stormflow and baseflow were not separated in this study) was summed across each period for each plot. 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, Growing 2013), treatment, and season*treatment interaction as fixed effects, and rep as random effect.

Results and Discussion

Soil:

Soil physical and chemical characteristics were more favorable for native plant growth on BROWN than either MIXED or GRAY (Table 2.1).

Table 2.1: Means and standard errors of soil pH, electrical conductivity, Mehlich-III-extractable P, K, Ca, Mg, and Zn, cation exchange capacity, texture classes, and exchangeable K, Ca, Mg, and Na. Means with the same letter are not significantly different ($p < 0.05$); means are compared within years.

	2005			2013		
Parameter	Brown	Gray	Mixed	Brown	Gray	Mixed
pH	6.03b ± 0.47	8.07a ± 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
P (mg kg^{-1})	6.09a ± 0.09	2.62b ± 1.0	0.53b ± 0.03	5.47a ± 0.84	2.47b ± 1.5	1.47b ± 0.03
K (mg kg^{-1})	51.1a ± 3.6	50.0a ± 5.8	52.5a ± 6.1	62.6a ± 1.0	27.6b ± 1.8	56.3a ± 5.2
Ca (mg kg^{-1})	573b ± 133	1080ab ± 292	1595a ± 28	653a ± 97	838a ± 465	887a ± 94
Mg (mg kg^{-1})	368a ± 8.8	295a ± 75.5	251a ± 10.5	304a ± 3.9	282a ± 104	274a ± 14.1
Zn (mg kg^{-1})	2.33a ± 0.04	3.77a ± 0.36	3.05a ± 0.76	2.78a ± 0.19	4.68a ± 0.45	6.34a ± 0.61
CEC (meq $100g^{-1}$)	8.27a ± 1.7	2.54b ± 0.38	3.38b ± 0.42	7.18a ± 1.3	3.28b ± 0.23	4.69b ± 0.50
% 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

Soil particle size distribution is one of the major differences among spoil types, with clay fraction highest 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 clays were highest on BROWN, silts were highest on MIXED, and rapidly rising, consistent with observations by Miller et al. (2012) that spoils containing shales weather into fines more rapidly.

We observed higher water holding capacity on BROWN than MIXED or GRAY, consistent with observations by Angel (2008) that soil moisture was higher 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 6-6.5. In contrast, soil pH on GRAY and MIXED was greater than 8.0 since construction, outside the growing range of most native hardwoods. Soil pH rose in all spoils from 2005-2007, 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.

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 2.2).

Table 2.2: Tree Volume Index (cm^3) and percent survival, with standard errors, by spoil type and year. 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 combined.

	2005			2007			2013		
	Brown	Gray	Mixed	Brown	Gray	Mixed	Brown	Gray	Mixed
Volume									
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 ± 1889	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 ± 2409	380c ± 147	4,183b ± 1280
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

Consistent with predictions from soil data, it is clear that BROWN provides a 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 MIXED or BROWN. This was somewhat anomalous, because it was clear at that point that tree volume was higher on BROWN than the other spoils. By 2013, however, survival on GRAY and MIXED had dropped below 70%, while survival on BROWN stayed at 86%. 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 are 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 2.3).

Table 2.3: 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

During plot construction, consistent with FRA recommendation 3 (minimize vegetative competition), no groundcover was seeded on the sites. However, 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 GRAY or MIXED all three 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 alum-root (*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., 2009).

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 three years (BROWN: 829 $\mu S\ cm^{-1}$; GRAY: 1032 $\mu S\ cm^{-1}$; MIXED: 1224 $\mu S\ cm^{-1}$); however, no significant differences were detected in the current sampling period, May 2012 through November 2013 (BROWN: 421 $\mu s\ cm^{-1}$; GRAY: 564 $\mu S\ cm^{-1}$; MIXED: 455 $\mu S\ cm^{-1}$) (Table 2.4).

Table 2.4: Means and standard errors of constituent concentration of water samples by spoil type during the sample period (March 2012 through November 2013). PROC MIXED found no significant treatment effects for any parameter ($p = 0.05$). BDL = below detectable limit.

Parameter	BROWN	GRAY	MIXED
EC (grab, μS cm-1)	421 ± 39	564 ± 55	455 ± 39
EC (remote, μS cm-1)	246 ± 16	366 ± 20	343 ± 14
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

Agouridis et al. (2012) found that EC, while initially much higher in MIXED and GRAY, was declining more sharply in these spoils than in BROWN and projected that they would be below the proposed 500 $\mu S\ cm^{-1}$ threshold by 2008. However, this rapid rate of leaching and weathering did not persist. Data after 2007 show that current EC trends are not significantly different from zero ($p = 0.05$), which suggests that EC has stabilized in all treatments (Table 2.5). Although no trends were apparent, EC readings were highly variable throughout the sampling period, in both flow-weighted and remote sampling techniques (Figure 2.1).

Table 2.5: Coefficients and intercepts (with standard errors) of constituent concentrations in water samples by spoil type (March 2012 through November 2013). Independent variable is days from first sample in current period.

	Treatment					
Constituent	Brown		GRAY		MIXED	
	Intercept	Slope	Intercept	Slope	Intercept	Slope
EC (grab)	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

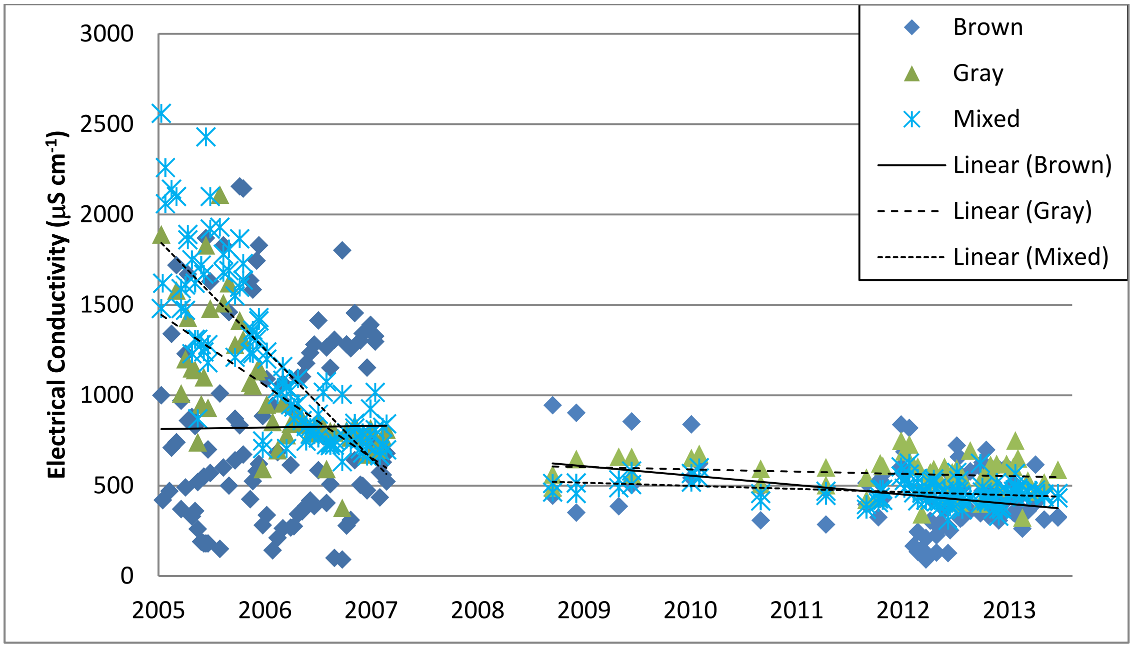

Figure 2.1: Electrical conductivity of flow-weighted samples, separated into sampling groups (2005-2007) and (2009-2013).

Interestingly, EC recorded by our *in situ* logger tended to be lower than EC from flow-weighted grab samples (BROWN: 246 vs. 421; GRAY: 366 vs. 564; MIXED: 343 vs. 455) (Figure 2.2).

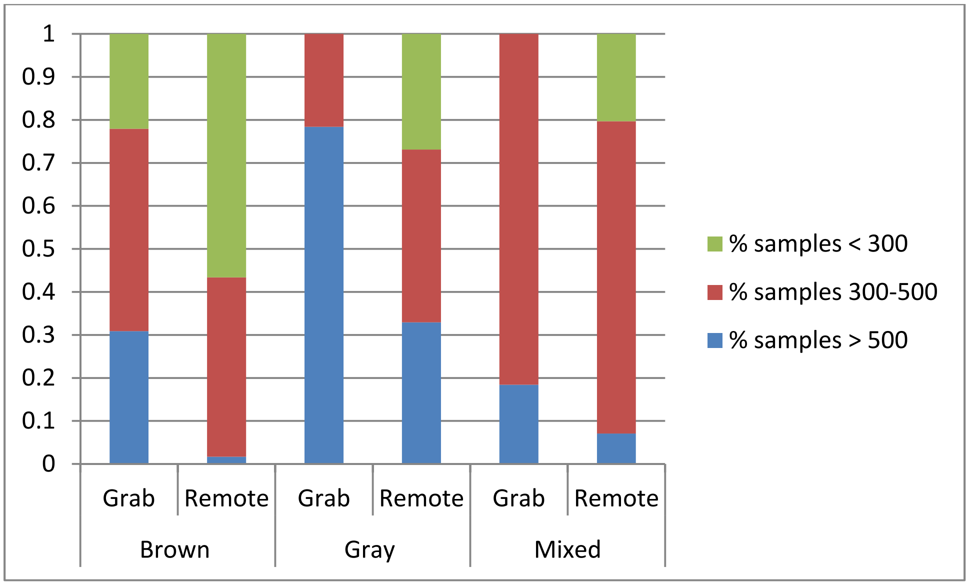

Figure 2.2: Electrical Conductivity (EC) measurements are categorized relative to macroinvertebrate toxicity thresholds (Pond et al., 2008). Percent of total samples falling into each toxicity category are compared between flow-weighted grab samples and *in-situ* logged samples as well as among spoil types (BROWN, GRAY, MIXED).

Remote EC readings were characterized by sharp spikes and dips (data not shown). Comparisons with rainfall records suggested that EC spiked initially after a rain event, eventually falling back to a pre-event level. These data suggest an initial flushing effect, in which interflow generated by the rain event initially dissolves and flushes salts and minerals that had been weathered since the last rain event. After flushing, EC drops abruptly, suggesting that increasing stormflow volume past a point results in a dilution effect. Of our sampling methods, remote monitoring was more sensitive to these storm-induced variations in EC, which may explain why average remote EC was consistently lower than average grab EC.

Although remote-monitored EC tended to be lower than grab EC, we observed similar trends in EC across spoil types from both methods. In both methods, BROWN and MIXED have the highest percentage of samples under the 300-500 $\mu S\ cm^{-1}$ range. GRAY has comparatively few samples under 300-500 $\mu S\ cm^{-1}$. These data, while statistically nonsignificant, suggest a possible difference in EC discharged from these spoil types. It appears, given relatively high EC in GRAY, that the ratio of unweathered material to weathered material may still be higher in GRAY, leading to higher concentrations of soluble salts available for leaching.

Agouridis et al. (2012) observed that one BROWN plot consistently discharged water with higher EC than the other BROWN plot, a trend which continued to some extent during the current period. They suggested that pyritic material was present in the BROWN spoil for that plot, leading to acidification and carbonate leaching. As seen in Figure 2.3, EC of water discharged from BROWN 3 was consistently higher than that discharged from BROWN 1.

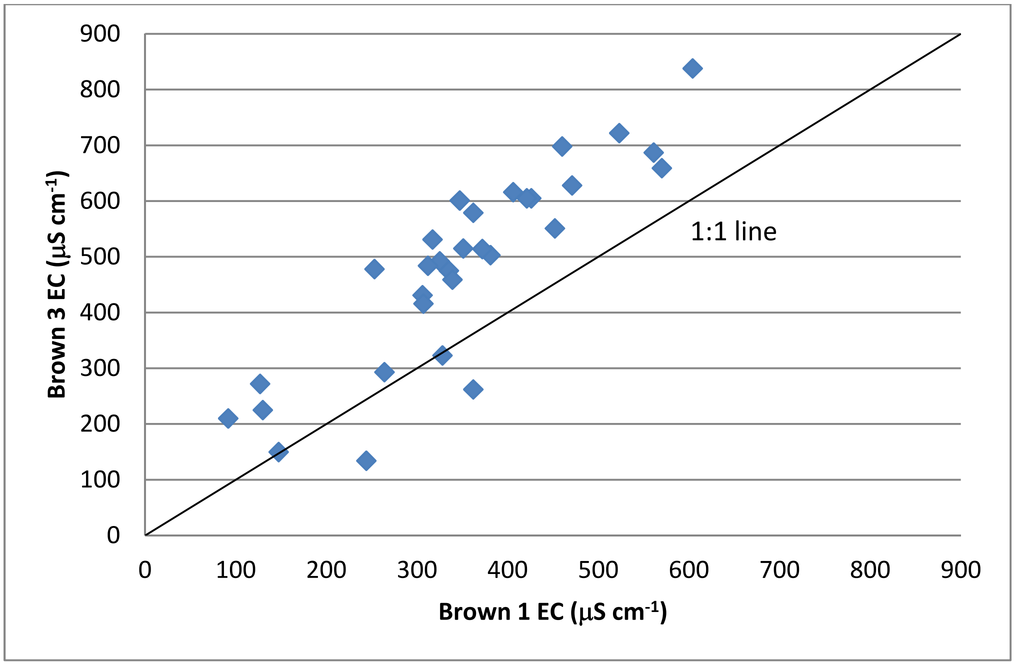

Figure 2.3: EC of flow-weighted grab samples from BROWN 3 compared to BROWN 1. A 1:1 line is plotted for reference.

This trend is consistent with early observations by Agouridis et al. (2012) and suggests that reactive materials continue to raise the EC on BROWN 3. Soil EC of minesoils derived from brown, weathered sandstones tends to be lower than soils derived from unweathered material (Zipper et al. 2013). Thus, it is speculated that if pyritic material can be handled appropriately, EC of water discharged from BROWN spoil would be even lower.

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 (Figure 2.4).

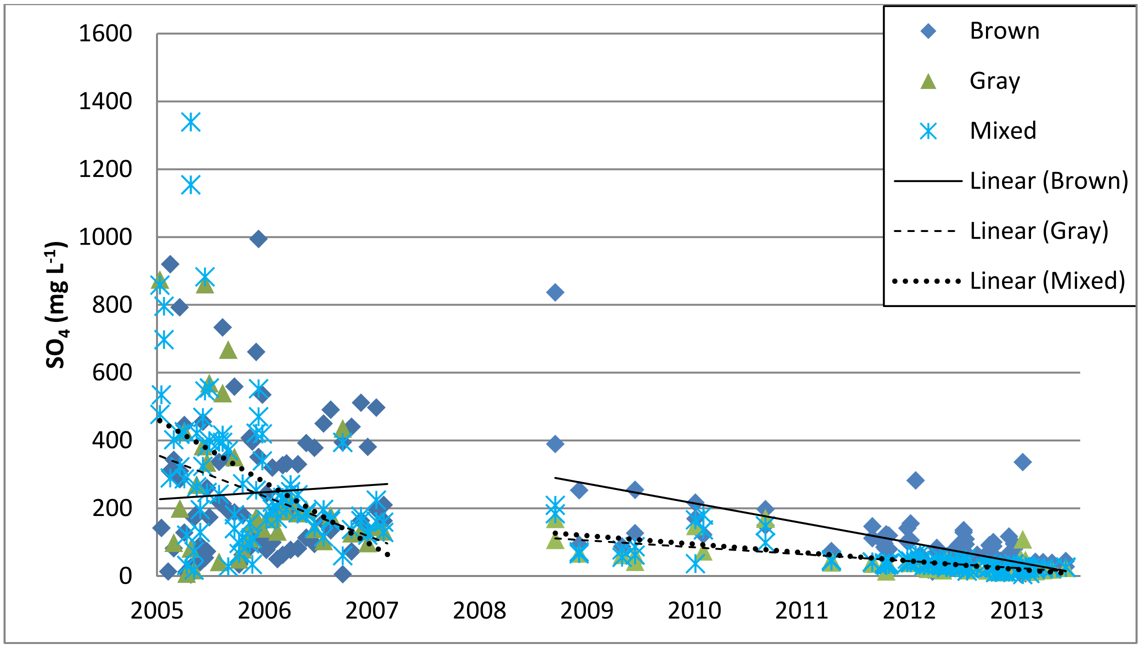

Figure 2.4: Sulfate concentrations in flow-weighted samples, separated into sampling periods 2005-2007 and 2009-2013.

During the current sampling period (2012-2013), 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 is declining in all treatments, but is lower in BROWN (294 mg L^{-1}) than GRAY (553 mg L^{-1}) or MIXED (431 mg L^{-1}) (Figure 2.5).

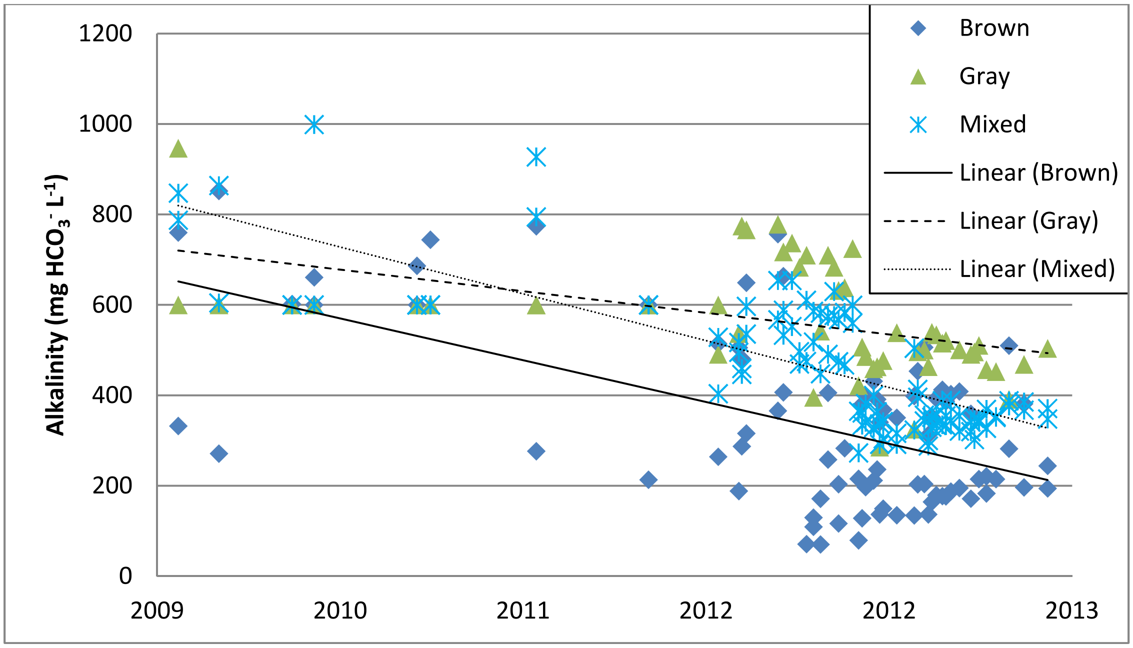

Figure 2.5: Alkalinity (as mg L^{-1} HCO_3^-) of flow-weighted samples, 2009-2013.

Declining alkalinity suggests that much of the rapidly dissolvable carbonate containing minerals have 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 2005-2007 and was equilibrated or nearing equilibration by 2013.

From 2005-2006, 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 GRAY (7.9) or MIXED (7.8), all lower than mean pH values in 2007 (Figure 2.6).

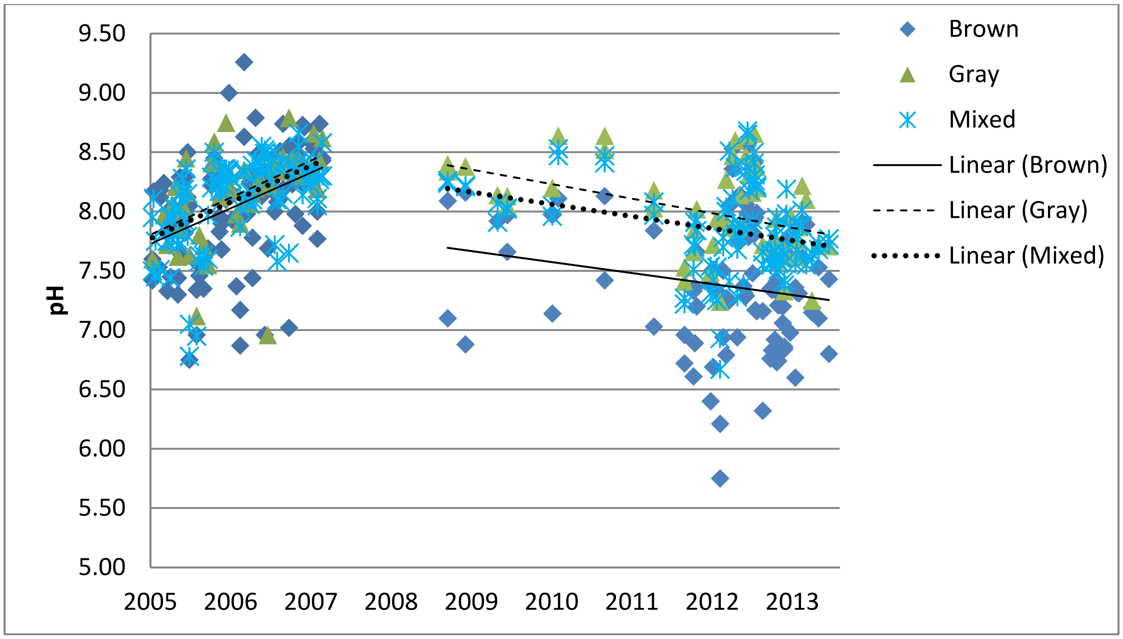

Figure 2.6: pH of flow-weighted samples, separated into sampling periods 2005-2007 and 2009-2013.

These results are consistent with decreasing alkalinity in suggesting that 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 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.

Overall, quality of water discharged from these sites was improved in 2013 over 2007. This demonstrates that placement of loose-dumped spoils during reclamation is followed by a period of rapid weathering, during which time EC will be quite high. Depending on specific spoil composition, additional constituents (e.g., heavy metals) may be of concern. However, this study shows that the pool of highly reactive minerals and salts can be rapidly depleted (3-9 years). By 2013, most constituents were present in stable concentrations, indicating that the systems had reached a steady state. It also appears that previously observed trends in soil suitability of weathered spoils over unweathered spoils holds generally true for interflow, with EC in BROWN nominally lower than GRAY. Of course, care must be taken to ensure that pyritic materials are effectively isolated from weathering to prevent sulfate spiking and high acidity. Consistent with early observations by Agouridis et al. (2012), we predict that streams draining sites reclaimed with weathered sandstones will benefit from lower concentrations of some constituents and lower overall EC; thus, they will suffer less biotic impairment.

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. (2009) observed that storm response on all treatments at Bent Mountain was similar to storm response in undisturbed reference forest. These observations suggest that minimizing 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. (2009) found only minor differences in hydrologic function among plots, but suggested that continued forest development would contribute to altered hydrologic function 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 hydrologic function among treatments was observed (Figure 2.7).

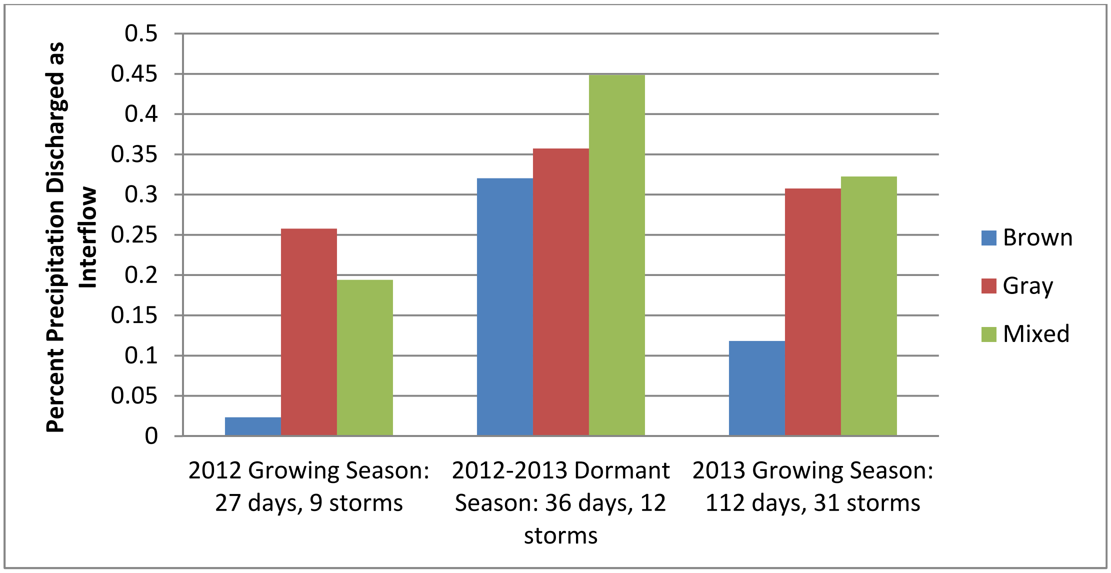

Figure 2.7: Discharged interflow was summed for each plot during periods for which accurate data were available for all plots. Interflow volume was expressed as a percentage of precipitation volume. Number of days and storms indicate the number of days and storms for which accurate data were available during each respective season.

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 on one of the GRAY plots (Angel, 2008), hydrologic 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. 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. It appears that seasonal variation was more extreme in BROWN than either GRAY or MIXED (Table 2.6).

Table 2.6: Percentage of rainfall discharged as interflow by plot by season. Percentages with the same letter (abc) are not significantly different down the column, while percentages with the same letter (xyz) are not significantly different across the row.

Season	2012 Growing	2012-2013 Dormant	2013 Growing
Brown	2.3a,x	32.0a,y	11.8a,x
Gray	25.8b,x	35.7a,x	30.8b,x
Mixed	19.4ab,x	44.9a,y	32.2ab,xy

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. These results indicate that seasonal variation in water budgeting is more extreme on BROWN than GRAY.

Given that vegetative cover and volume are so much greater on BROWN than either MIXED or GRAY, an evapotranspiration effect would be expected. Discharge on BROWN is 20-30 percentage points lower in the growing season than in the dormant season, compared to 5-10 points in GRAY and 13-25 points in MIXED. We propose that this more extreme seasonal decrease in the amount of interflow discharged is mediated by increased transpiration in BROWN. If tree and groundcover growth on BROWN continue to outpace growth on GRAY and MIXED, we expect that this difference will become more apparent.

This study is the first to demonstrate that developing forest on mined land contributes significantly to water budgeting. If appropriate reforestation techniques (e.g., FRA) are implemented and weathered spoils are used in place of unweathered spoils when soil substitutes are required, forest development and subsequent transpiration effects may be more extreme. 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. These observations suggest that long-term forest development on mined sites may be able to contribute to mitigation of downstream water quality impacts by reducing the volume of water discharged from reclaimed sites. 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.

Conclusions

After nine growing seasons, it appears that BROWN unweathered spoils provide an improved growth medium for native trees and a reduced threat to local aquatic communities. Overall plant preference for BROWN was demonstrated by significantly higher tree volume and groundcover in BROWN than 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 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 has 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 three 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 equilibration after only nine growing seasons. Compared to observations of conventionally reclaimed sites still contributing significant contamination to streams ten years after reclamation (Fritz et al., 2010), our study shows that water quality from spoils placed according to FRA recommendations rapidly improves (within nine years) to acceptable levels for downstream aquatic communities (e.g., EC from 300 to 500 $\mu S\ cm^{-1}$).

While no significant differences were detected for water quality parameters, BROWN spoil discharged water with the lowest mean EC, pH, and alkalinity, suggesting that BROWN spoil may be a preferable soil substitute for reducing long-term water quality impacts. In addition to discharging slightly less contaminated water, improved tree growth and evapotranspiration on BROWN contributes to lower discharge during the growing season, suggesting that areas reclaimed using BROWN weathered spoils and planted with native hardwoods will have reduced impact on local and downstream aquatic ecosystems after nine growing seasons. Follow-up studies will be necessary to document if current trends in seasonal variance in hydrologic function and slightly improving water quality continue over the next few years.

Appendix A: Species composition by spoil type, 2013.*

Latin Name	Common Name	Type	Native	Brown	Gray	Mixed
Acer negundo	Boxelder	T	Yes	+	0.7	-
Acer rubrum	Red maple	T	Yes	0.4	0.5	1.2
Ageratina altissima	White snakeroot	H	Yes	-	+	-
Ailanthus altissima	Tree of heaven	T	No	2.3	+	1.3
Andropogon sp.	Broom-sedge	G	Yes	1.9	60.8	60.6
Apocynum cannabinum	Indianhemp	H	Yes	-	-	+
Betula lenta	Sweet birch	T	Yes	0.7	+	+
Buddleja davidii	Butterfly bush	T	No	-	0.4	2.3
Campsis radicans	Trumpet creeper	V	Yes	-	-	+
Cardamine pennsylvanica	Pennsylvania bittercress	H	Yes	-	5.4	-
Cercis canadensis	Eastern redbud	T	Yes	1.4	-	+
Chenopodium album	Lambsquarter	H	No	-	+	-
Cirsium sp.	Thistle	H	No	0.2	-	+
Clematis virginiana	Virgin's bower	V	Yes	0.1	0.4	1
Cornus drumondii	Roughleaf dogwood	T	Yes	+	-	-
Cornus florida	Flowering dogwood	T	Yes	+	+	+
Desmodium sp.	Ticktrefoil	H	**	-	0.4	+
Eleagnus umbellata	Autumn-olive	T	No	8	-	0.6
Erigeron sp.	Fleabane	H	Yes	0.1	3.4	1.8
Eupatorium fistulosum	Hollow stem Joe-pye weed	H	Yes	+	-	-
Eupatorium serotinum	Lateflowering thoroughwort	H	Yes	+	-	1.3%
Eurybia divaricata	White wood aster	H	Yes	+	-	-
Festuca arundinacea	KY 31 fescue	G	No	-	2.5	0.9
Heuchera sp.	Alum-root	H	Yes	0.1	-	-
Hydrangea sp.	Hydrangea	T	Yes	+	-	+
Juncus sp.	Rush	G	**	1.3	-	-
Juniperus virginiana	Eastern redcedar	T	Yes	+	-	1.8
Lactuca saligna	Willow-leaf lettuce	H	No	-	0.4	+
Lactuca serriola	Prickly lettuce	H	No	0.9	0.4	+
Lamiaceae	Mint family	H	**	+	-	-
Lespedeza cuneata	Sericea lespedeza	H	No	42.8	4.3	11.4
Lespedeza striata	Kobe lespedeza	H	No	1.8	+	0.4
Lobelia inflata	Indian-tobacco	H	Yes	-	-	-
Lonicera japonica	Japanese honeysuckle	T	No	0.1	-	+
Ludwigia leptocarpa	Anglestem primrose-willow	H	Yes	-	-	0.8
Microstegium sp.	Japanese stiltgrass	G	No	0.7	-	-
Miscanthus	Silvergrass	G	No	1.8	4.1	0.4
Packera anonyma	Small's ragwort	H	Yes	-	0.7	0.4

Parthenocissus sp.	Virginia creeper	V	Yes	0.3	+	0.4
Paulownia tomentosa	Royal paulownia	T	No	9.8	0.4	6.7
Phytolacca americana	Pokeweed	H	Yes	+	1.4	+
Pinus echinata	Shortleaf pine	T	Yes	-	-	+
Pinus strobus	White pine	T	Yes	-	-	+
Pinus virginiana	Virginia pine	T	Yes	0.3	+	+
Plantago major	Common plantain	H	No	-	-	-
Platanus occidentalis	Am. Sycamore	T	Yes	1.7	0.4	0.4
Poa pratensis	Kentucky bluegrass	G	No	0.5	-	-
Polygala sanguinea	Purple milkwort	H	Yes	1.7	2.2	0.9
Polystichum acrostichoides	Christmas fern	H	Yes	+	-	-
Populus deltoides	E. cottonwood	T	Yes	+	-	-
Populus grandidentata	Bigtooth aspen	T	Yes	+	-	-
Prunus serotina	Black cheery	T	Yes	+	+	-
Rhus typhina	Staghorn sumac	T	Yes	2.7	3.2	+
Robinia pseudoacacia	Black locust	T	Yes	13.3	0.7	-
Rosa multiflora	Wild rose	T	No	1	-	-
Rubus allegheniensis	Allegheny blackberry	T	Yes	4.5	+	3.1
Rubus flagellaris	Northern dewberry	H	Yes	+	-	-
Rubus occidentalis	Black raspberry	T	Yes	0.3	0.4	-
Rubus phoenicolasius	Wineberry	T	No	+	0.7	-
Salix nigra	Black willow	T	Yes	0.2	-	-
Sassafras albidum	Sassafras	T	Yes	0.7	1.4	-
Solidago canadensis	Canada goldenrod	H	Yes	3.6	3.8	1.5
Symphoricarpus orbiculatus	Coralberry	T	Yes	+	-	-
Symphotrichum sp.	Aster	H	Yes	0.5	6.9	2.1
Taraxacum officinale	Common dandelion	H	No	0.3	1.1	-
Toxicodendron radicans	Poison ivy	V	Yes	+	+	+
Tussilago farfara	Coltsfoot	H	No	0.1	3.4	6.8
Ulmus americana	American elm	T	Yes	0.2	+	+
Verbascum thapsus	Common mullein	H	No	+	+	-
Verbena urticifolia	White vervain	H	Yes	+	-	-
Vernonia gigantea	Giant ironweed	V	Yes	+	+	-
Viburnum dentatum	Southern arrowwood	T	Yes	-	-	0.4
Vitis sp.	Wild grape	V	Yes	0.5	4.3	+

+ = Species observed on indicated spoil type, but not detected by Farmer method (Farmer et al. 1981)
- = Species not observed on indicated spoil type

*Percentages reflect amount of total naturally regenerated cover on indicated spoil attributed to indicated species
**Native status unknown
T = tree/shrub, V = vine, H = herb, G = grass

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