POWELL RIVER PROJECT

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PROGRAM REPORTS

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Table of Contents

POWELL RIVER PROJECT

A Land-Grant University Program Serving Southwestern Virginia's Coalfield Counties

Overview of Powell River Project

Programs

Research and Education Center

Extension Publications

Research Results

Research and Education Program Reports

Accomplishments

Contacts Links Sponsors

Virginia Cooperative Extension

A Study to Determine the Preference of Nesting Box Entrance Hole Size of *Sialia sialis* (Eastern Bluebird)

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Nesting boxes with large entrance holes were paired with boxes with the traditional entrance hole size at the Powell River Education Center to test whether the Eastern Bluebird (*Sialia sialis*) would utilize the boxes with the larger opening. During the 2010 nesting season (mid-April to mid-August), nesting activity was observed in both box types. Bluebirds, tree swallow and chickadees were active in field 1. Bluebirds and tree swallows successfully fledged young. One chickadee nest was lost due to predation, but at the time of the writing of this report, a chickadee nest in a large entrance hole box was active with chicks. Bluebirds and chickadees were active in field 2, however both nest failed in fledging young and all nesting activity ended in the field by mid-June when the grass in the field grew tall. During 2010 nesting season, a total of 15 bluebirds fledged and possibly 3 chickadees will fledge; the one successful tree swallow nest fledged 5 chicks. Student volunteers from the Mountain Empire Community College assisted in the removal of open-topped boxes from the previous study, modification and installation of new boxes and the monitoring of nesting activity.

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Introduction

During the fourth year of the project along the bluebird trail established at the Powell River Education Center, open-topped boxes were removed and replaced with closed topped boxes with enlarged entrance holes in order to test whether Eastern Bluebirds, when given a choice, had a preference in entrance hole size. The round hole diameter specified by the North American Bluebird Society, and used in most commercially produced bluebird boxes, is 1.5 inches (www.nabluebirdsociety.org/nestboxspecs.htm). Bermudez (2002) published a study in which he presented data demonstrating that Eastern Bluebirds would utilize and successfully fledge young from boxes with entrance holes enlarged to a diameter of 2.75 inches. Competitors such as European Starlings and House Sparrows have been found to avoided boxes with large entrance holes (McGilvrey and Uhler, 1971; Heusmann et. al., 1977).

Student volunteers from Mountain Empire Community College assisted in trail maintenance, the removal of open-topped boxes from the previous study, the installation of new boxes and the monitoring of nesting activity. B. Wells modified the boxes.

Methods

Entrance hole preference- Nesting boxes of the Bermudez design were paired with boxes with the traditional entrance hole diameter along the Bluebird trail established at the Powell River Education Center (figure 1a, b). The standard bluebird box has an entrance hole 1.5 inches in diameter (Figure 2a), while the Bermudez box has a hole diameter of 2.75 (Figure 2b). Nesting boxes were monitored for activity on a weekly basis beginning April 13 through the submission date of this report. Monitoring activity followed the protocols established by the North American Bluebird Society (Fact Sheet: Monitoring Bluebird Nest Boxes, 2002) and the Virginia Bluebird Society (Virginia Bluebird Trail Monitoring Information, 2004). Data was recorded on forms provided on the Virginia Bluebird Society website.

Figure 1(a). Nesting box sites in field 1 and (b) field 2. Numbers indicate the box locations. Arrow indicates north. The B indicates the position of the barn. Yellow lines indicate the location of the main road. (Image from Microsoft Virtual Earth.)

Figure 2. (a) Nesting box with 1.5 inch diameter entrance hole and (b) nesting box with 2.75 inch diameter entrance hole.

Survey of insect and invertebrate populations. During the breeding season, insects and other invertebrates make up approximately 68% of the Eastern Bluebird diet (Fimbel, 2006) and include: grasshoppers, crickets, katydids, beetles, earthworms, spiders, millipedes, sow bugs, and snails [Eastern Bluebird (*Sialia sialis*), 1999]. Insects and other invertebrates were sampled at 4 sites in field 1 only (figure 3), due to the paucity of nesting activity in field 2. Samples were collected using (1) passive pan traps that were in place for seven days and captured both flying insect and organisms that live at ground level (Terrestrial Arthropod Densities, 1994), and (2) insect nets that collected insect by sweeping the grass along a 30 m x 1 m transects (Perry et al, 2001). Pan traps were placed outside the fences in case cattle were moved into the field during the sampling period. Pan traps [13 in x 9 in metal cake pans sprayed yellow (Terrestrial Arthropod Densities, 1994)], were placed flush with the substrate, and filled with a soap and salt solution. The soap and salt solution acted as a trap and as a temporary preservative. Specimens were collected from the traps by pouring the contents though a strainer. The specimens were rinsed and placed in 95% ethanol. Transect sampling took place inside or along the fence lines where the grass was short and bluebirds would most likely hunt. Bluebirds prefer areas where the grass is short and perches are available to spot their prey (www.bluebirdsforever.com/trail.html). Animals collected along transects were also transferred to jars containing 95% ethanol. Specimens were identified and sorted into groups using the National Audubon Society Field Guides to North American Insects and Butterflies.

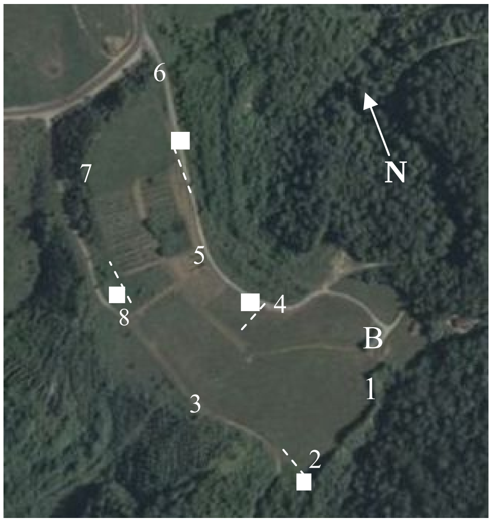

Figure 3. Field 1: Nesting box sites indicated by numbers. Pan sample locations indicated by □'s. Transect locations indicated by dashed lines (----). Arrow indicates north. The B indicates the position of the barn. (Image from Microsoft Virtual Earth.)

Results

Nesting activity: Species active along the trail during the 2010 season included bluebirds, tree swallows and black-capped chickadees (Table 1). The first eggs were found on April 13 in boxes with the small entrance at sites 3, 7 and 12 (5, 2 and 5 eggs respectively). Nest building activity was also noted in box 5 A (small entrance hole) on that date. One addition egg had been laid in box 7A, while 3 additional eggs were laid in box 12A by the following week. By April 30, 3 of the 5 eggs had hatched in box 3A, 4 eggs had hatched in box 7A as well as 12A. By May 13, the 6 chicks in box 7A were close to fledging, while the chicks in boxes 3A and 12A had disappeared. The chicks in those two nests were too young to fledge and were assumed lost to predation. Additional activity was detected in box 12A on June 2; a chickadee nest had been constructed on top of the bluebird nest. No eggs were ever laid in the nest, and by the following week, the nest was infested with black ants and removed. No further activity was found in any of the boxes in field 2.

The first nesting activity in box 6A was found on May 13; however, the top had been knocked off the box and no eggs were found in the nest. On the same date, a nest with 4 bluebird eggs was found in box 6B, the box with the large entrance hole, but by the following week the eggs were gone. No further activity occurred in either box for the remainder of the breeding season. Three and five eggs, respectively, were found in boxes 4A and 5A on May 21. The bluebird in box 4A eventually fledged 5 chicks, while the tree swallows in box 5A also fledged 5 chicks. Bluebirds were again active in box 7A on June 18, chicks were in the nest by July 7, but were gone by the following week and assumed lost to predation. Five tree swallow eggs were found in box 3A on June 2, but the eggs were lost to predation.

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Box	Species	Nest building	# of Eggs	# of Hatchlings	# Fledged
1 A	CH	Yes	0	0	0
1 B	---	No	0	0	0
2 A	---	No	0	0	0
2 B	---	No	0	0	0
3 A	BB;TS	Yes	5; 5	3; 0	0
3 B	---	No	0	0	0
4 A	BB	Yes	5	5	5
4 B	---	No	0	0	0
5 A	BB; TS	Yes	9; 5	9; 5	9; 5
5 B	---	No	0	0	0
6 A	?	Yes	0	0	0
6 B	BB	Yes	4	0	0
7 A	BB; TS	Yes	10; 0	6; 0	6; 0
7 B	CH	Yes	3?	3?	Box still active
8 A	---	No	0	0	0
8B	---	No	0	0	0
9 A	---	No	0	0	0
9 B	---	No	0	0	0
10 A	CH	Yes	5	5	0
10 B	---	No	0	0	0
11 A	---	No	0	0	0
11 B	---	No	0	0	0
12 A	BB	Yes	5	4	0
12 B	---	No	0	0	0
13 A	---	No	0	0	0
13 B	---	No	0	0	0

Table1. Nesting results for the 2010 nesting season. (A: small box hole; B: large box hole; BB: bluebirds; CH chickadees; TS tree swallows).

Nesting activity was again observed in box 5A on July 7, and 4 bluebird eggs were found in the nest on July19. Three of the 4 eggs hatched by July 26, while the remaining chick hatched by the next week and all 4 chicks fledged by August 9. Also on July 7, nesting activity was detected in box 7B. By July 19, 3 chickadee eggs were found in this large entrance box. The 3 chicks hatched by August 2, and were still in the nest at the writing of this report.

Insect and invertebrate survey: Insects and other invertebrates were sampled at four sites by insect net [July 26 (Table 2)] and by pan traps [July 19-26 (Table 3)]. A total of 3301 specimens were identified in these samples. The largest numbers of specimens were collected by the pan traps. Specimens were identified and placed into one of twenty-five invertebrate groups using Milne et al. (2005).

30 m Transects

Group / Sample	Sample Location (between boxes) 2-3	4-5	5-6	7-8
Ants	1	1	1	1
Aphids	0	1	0	0
Bees	6	20	56	31
Beetles	8	12	10	8
Butterflies	0	0	0	0
Caterpillars	0	0	0	2
Centipedes	0	0	0	0
Crickets	4	3	7	1
Dragonflies	0	0	0	0
Earwigs	0	0	0	0
Flies	4	28	51	11
Grasshoppers	1	3	12	2
Lacewing	0	0	0	0
Leafhoppers	58	132	142	46
Long-necked seed bug	0	4	6	1
Mosquitoes	0	2	4	4
Moths	0	0	0	0
Roach	0	0	0	0
Sawflies	0	1	1	0
Slugs	0	0	0	0
Snails	0	0	0	0
Spiders	17	16	10	28
Ticks	0	0	0	0
Wasps	2	0	0	6
Weevils	2	0	0	2

Table 2. Results of insect and invertebrate transect surveys conducted July 26, 2010.

Pan Traps

Group / Sample	Sample Location (between boxes) 2-3	4-5	5-6	7-8
Ants	5	43	17	10
Aphids	0	0	0	1
Bees	31	247	175	55
Beetles	16	30	30	29
Butterflies	2	9	8	0
Caterpillars	4	1	1	0
Centipedes	0	0	3	1
Crickets	2	7	15	9
Dragonflies	1	1	0	0
Earwigs	4	10	14	15
Flies	18	100	191	40
Grasshoppers	5	12	20	26
Leafhoppers	124	332	317	133
Long-necked seed bugs	0	12	4	0
Mosquitoes	2	5	3	1
Moths	21	72	28	5
Plant hopper	1	0	3	0
Roach	0	0	0	1
Sawflies	8	22	20	3
Slugs	0	0	0	2
Spiders	12	35	24	26
Stink bugs	0	2	3	0
Ticks	3	20	5	0
Wasps	1	12	17	21
Weevils	1	5	3	2
Yellowjacket	1	13	0	0

Table 3. Results of insect and invertebrate pan surveys conducted between July 19 and July 26, 2010.

Discussion

Bermudez (2002) found that bluebirds would utilize nesting boxes with entrance holes enlarged from a diameter of 1.5 inches to 2.75 inches. The objective of this study during the 2010 season and during the 2011 season, is to test whether bluebirds will show a preference for one of the two entrance hole sizes when presented with both. During the course of the 2010 season, nesting activity was observed in two of the large entrance boxes. Bluebirds built a nest and produced 4 eggs in box 6B; however, it is possible that they would not have used this box if box 6A had not lost its top exposing the nest. Chickadees built a nest in box 7B late in the season, and unlike the bluebirds have to this point avoided predation. This box is within a few feet of the tree line and shaded by an over hanging evergreen branch. The location of the box may have obscured the box from predators and kept the box cool in the high summer temperatures [+90° F (personal observation)]. This study will be continued in the 2011 season, to determine if birds continue to utilize these boxes.

The composition of the insect populations changed from the previous season. The preferred food items for bluebirds [beetles, butterflies, crickets, grasshoppers, leafhoppers moths and spiders (All About Birds- Eastern Bluebird, 2003)] during the 2009 season ranged from 49.9% to 57.3% of the groups present in the combined tallies at each sampling site. During the 2010 season, the preferred food species increased to 75.4% of the identified specimens, ranging from 71.1% to 84.3% in the combined tallies at each sampling site

The bluebird trail at the Powell River Educational Center successfully supported the reproductive efforts of Eastern Bluebirds, chickadees and tree swallows. Both bluebirds and chickadees were found to utilize the Bermudez boxes. In the 2011 season, it will be interesting to determine if the birds will continue to utilize the Bermudez boxes or whether the two cases form this season were isolated incidences.

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Wildlife Response to Surface Mine Reclamation in Southwest Virginia

Chris E. Latimer and Dean F. Stauffer

Introduction

Post-mined lands can be managed for a number of different land uses including managed pasture for grazing livestock, commercial forest products, outdoor recreation, and carbon sequestration. No matter what the post-mining land use is, there will always be opportunity to manage the land to provide suitable wildlife habitat. For example, several studies suggested that the new environments created by strip mining are beneficial to wildlife not common to the area or that are suppressed by limited resources and competition (Allaire 1978, Whitmore and Hall 1978, Rohrbaugh and Yahner 1996 Bajema et al. 2001). Much of the area affected by coal mining in Virginia is forested ridgetops which provide important nesting habitat for species of concern such as the cerulean warbler (*Dendroica cerulea)*; however, reclaiming strip mines creates large amounts of early successional habitat used by many other species that are currently in decline such as the field sparrow (*Spizella pusilla)*, golden-winged warbler (*Vermivora chrysoptera)*, and yellow-breasted chat (*Icteria virens)* (Carrozzino 2009). Therefore, reclaimed strip-mines provide a unique opportunity to study habitat use by birds and identify critical components of the habitat that are important for management.

Loss of breeding habitat due to fragmentation has been hypothesized as a major cause of population declines in forest-nesting Neotropical migrant birds (Donovan and Flather 2002). Surface mining activities can also lower the structural diversity of various cover types and may lead to structural and compositional changes in avian communities (Wray et al. 1982). Changes in cover types could also cause a change in the predator community and thereby alter predator-prey dynamics. Sedimentation from erosion can impact aquatic habitats and the various vertebrate and invertebrate communities associated with them. Other environmental issues that may impact wildlife populations on surface mines include noise, air quality, construction and use of temporary road systems, and environmental contamination.

Many studies have focused on the negative impacts of surface mining activities on wildlife; however, few studies have assessed the value of post-mined lands as wildlife habitat and provide management recommendations for the development of suitable habitat for wildlife (Carrozzino 2009). Carrozzino (2009) found a predictable trend between breeding bird species richness, and abundance and successional stage on reclaimed surface mines. While her study confirmed some value of reclaimed mine-lands for wildlife, species richness and abundance may not be reliable measures of habitat quality in all cases (VanHorne 1983). An understanding nest-site selection and nesting success is important to understanding the ecology and evolution of species and to develop management recommendations (Rodewald 2004). Such knowledge will contribute to our understanding of the contribution of reclaimed mine lands to sustaining populations of bird species of concern.

The primary focus of this study is to determine avian breeding success of birds nesting on reclaimed mine sites, specifically focusing on sites reclaimed using the Forest Reclamation Approach (FRA)(Burger et al. 2005). The continued monitoring of avian populations will contribute to the data collected by Carrozzino (2009) in order to evaluate changes in avian community structure over a 5-year period as the reclaimed sties mature. In addition, we will collect data on the use of reclaimed mine sites by medium and large sized mammals. Lastly, we will sample arthropods to evaluate seasonal and site-specific changes in arthropod community composition and biomass.

Research Objectives

This research is intended to contribute to our understanding of wildlife use of reclaimed mine lands. The specific objectives are:

1) Assess breeding bird community composition in various cover types reclaimed for different post-mining land uses on Powell River Project.
2) Monitor breeding success of songbirds (with emphasis on species of concern in Virginia) in varying mine reclamation regimens specifically focusing on sites reclaimed to hardwoods and are FRA compliant.
3) Determine what environmental factors may influence breeding success of songbirds in different cover types.
4) Model finite rate of increase of common songbird populations on the Powell River Project, using survival estimates from other studies in the region and our estimates of nest success to determine if reclaimed strip mines are acting as population sinks
5) Assess mammal community focusing on potential avian nest predators at The Powell River Project to determine its use of reclaimed mine sites and relate distribution and abundance to habitat conditions.
6) Relate information gained in this study to previous information about wildlife management on reclaimed mine sites to make recommendations about reclaiming lands for wildlife use.

Overview of Methods

Field work was conducted at the Powell River Project site in Wise County, Virginia, and surrounding reclaimed surface mines in Appalachia, Dickenson and Russell Counties. We have completed one year of data collection on this two year study. Sites for this study were chosen from patches of relatively homogeneous vegetation cover of similar age, reclaimed under the same technique (pre-SMCRA, FRA, grassland/pasture). Because of the relatively recent development and application of the Forest Reclamation Approach, availability of FRA sites is limited and most FRA sites are <20 years old. Therefore, in an effort to increase sample size, we also sampled sites that were FRA compliant. Because the maximum age of existing FRA site is <20 years of age, we only consider sites that have been reclaimed or undergone natural succession within the past 20 years. Because we are evaluating differences in patches reclaimed using different techniques, pre-SMCRA sites (no active reclamation) were sampled to collect baseline data. We located as many patches as possible on the 2 study areas that were large enough (3-4ha) for sampling and met the above constraints. Some patches were large enough to contain multiple sampling points; therefore, the number of sampling points was appropriate to the size of the patch sampled such that each point was >200m apart (Table 1).

Table 1. Number of points and location of study sites in Wise and Dickinson Counties, VA, 2010.

Site	# of points	Location
Powell River	32	Wise
PALS	7	Dickenson
Mud-Lick Creek	4	Appalachia
TNC Flint Gap	4	Russell

Bird Sampling

Survey points in pre-SMCRA, pasture and traditional reclamation cover types were adopted from Carrozzino (2009) and additional points were located within FRA and FRA-compliant cover types. We used variable radius point counts to record bird species and distance from point center to each individual bird. Sex and age class (if known) were also recorded for each individual. Each point was visited 4 times between May 18 and Aug 9. The first and last counts of the sampling period lasted a total of 5 minutes with a 1-minute “settling period” after arriving at each point. The 2nd and 3rd counts lasted a total of 10 minutes without a 1-minute settling period. All surveys were conducted between 6 and 9am on clear mornings with minimal wind. Observers and time of day were alternated between counts to minimize any biases that may occur.

Nests were located from late April to mid July (Figure 1) by using behavioral cues and systematically searching potential nesting substrates. A total of 58 nests from 18 different species were located in various cover types on reclaimed mine-lands (Table 1, Figure 2). Nests were checked every 2 to 3 days to monitor the contents and status of the nest. Steps were taken to minimize disturbance around the nest and to avoid potentially leading predators to the nest. A pole with a mirror was used to check the contents of nests located higher in the canopy as well as in the shrub layer to minimize disturbance.

Habitat measurements were taken within 10 days of the successful completion (at least one young fledges) or failure of a nest. Measurements were taken at 2 different spatial scales at used and unused nest sites. Unused sites were determined by choosing a random compass bearing and pacing 30m to the nearest nesting substrate of the same plant species as the used nest-site. At the immediate area surrounding the nest within a 1m radius, we measured the following habitat characteristics: nest height (m), nest substrate (spp. and DBH if applicable), distance from center of nest to center of substrate, distance from the center of the nest to the edge of the substrate, and distance of nest to nearest edge (the junction between two different cover types, or the same cover type with visible differences in age). Each edge was scored on a scale from 1 to 5, with 1 representing little contrast between 2 patches (same cover type, different age) and 5 representing the greatest contrast (e.g., junction between forest and pasture). Percent concealment of the nest was measured by visually estimating the percentage of the nest that was constructed by vegetation from 1m in each cardinal direction, above and below and then averaged to obtain an estimate of the total concealment.

Habitat measurements were also taken at the patch scale, defined as the area encompassed by a 0.04ha circular plot centered on the nest. At this scale, we measured percent canopy cover, canopy height, density of trees > 10 cm diameter at breast height (DBH), density of stems > 4 and < 10cm DBH (saplings/poles), and density of stems < 4cm DBH (shrubs). An estimate of understory foliage volume was obtained by using a standard vegetation profile board (Nudds 1977) divided into 5 height intervals, 0-0.5m, 0.5-1m, 1-1.5m, 1.5-2m and 2-2.5m. The board was 2.5m tall and 30.48cm wide with alternate black and white colors at 0.5m intervals. The proportion of each 0.5m interval covered by vegetation was be recorded on a scale from 1 to 6, corresponding to the 6 cover classes presented in Daubenmire (1959). Readings were taken 10m in each cardinal direction from the nest at a height of 1m and the average for each 0.5m interval will be used in analyses. Percent ground cover by category (grasses, forbs, woody vegetation <1m, moss, leaf litter), percent bare ground, rocks, and percent coarse woody debris (CWD, woody vegetation > 8cm DBH and >1m long) were also taken by using the point intercept method.

Table 2. Number of nests by species located on reclaimed mine lands in Wise and Dickinson Counties, 2010.

Common name	Species code	# Nest
Black-and-white warbler	BAWW	2
Blue-gray gnatcatcher	BGGN	1
Blue jay	BLJA	2
Brown thrasher	BRTH	2
Cedar waxwing	CEWA	1
Chipping sparrow	CHSP	1
Field sparrow	FISP	17
Gray catbird	GRCA	2
Golden-winged warbler	GWWA	1
Hooded warbler	HOWA	1
Indigo bunting	INBU	15
Northern cardinal	NOCA	1
Prairie warbler	PRWA	3
Red-eyed vireo	REVI	2
Red-winged blackbird	RWBB	1
Grasshopper sparrow	GRSP	2
White-eyed vireo	WEVI	2
Wild turkey	WITU	1
Yellow-breasted chat	YBCH	1
Totals	19	58

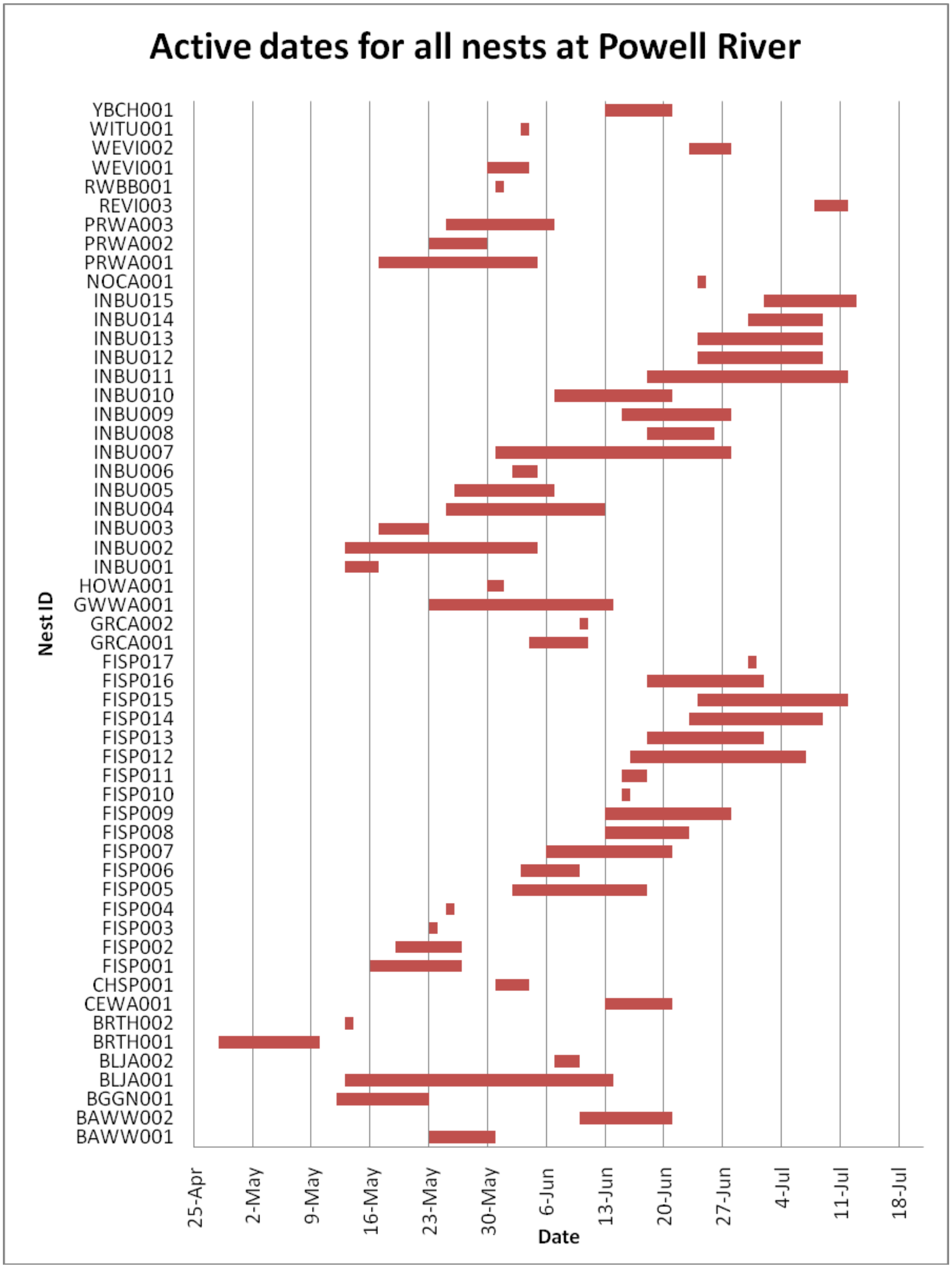

Figure 1. Active dates for all nests on the Powell River study site. Nests were active from April 28 through July 13, 2010. Species codes can be found in Table 1.

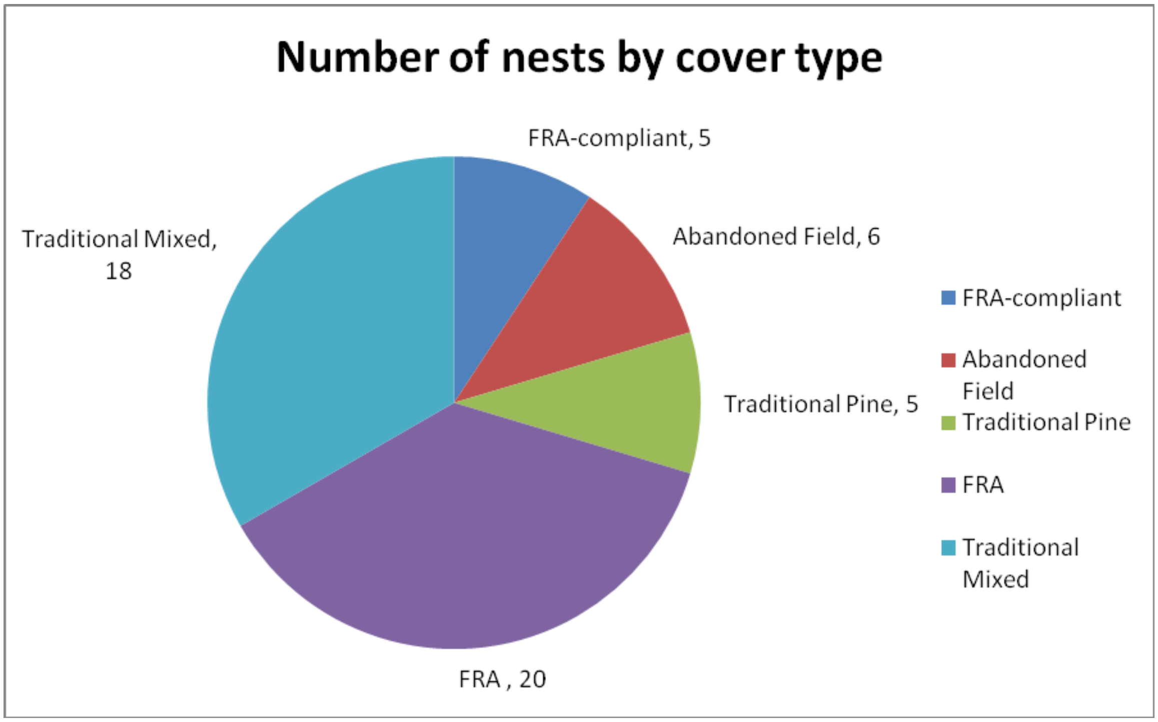

Figure 2. Number of nests located in cover types sampled in Wise and Dickinson Counties, VA, 2010.

Mammal sampling

Remote sensing camera-traps were used to inventory mid- to large-sized mammals in different cover types at the Powell River site. Cameras were spaced approximately 1km apart in areas of suspected animal activity (based on scat, trails, and tracks). A total of 9 camera-traps were deployed from June 5 through July 13, totaling 38 trap-nights per camera for a total of 342 trap-nights for the entire study area. Camera-traps were taken down early as a result of theft on the Powell River Project. Animal activity was calculated as the number of independent capture events for a single species divided by the total number of trap nights, and relative animal activity was calculated as the number of independent capture events for a single cover type divided by total capture across all cover types (Figure 3, Table 2). Consecutive photos of the same species were considered independent events if individual animals can be unambiguously identified, or if the interval between capture events was >30 minutes (Michalski and Peres 2007).

Table 3. Relative animal activity by cover type. Numbers expressed as # animals/# trap nights.

Relative Animal Activity								
	Black Bear	Bobcat	Coyote	Gray Squirrel	Raccoon	White Tailed Deer	Wild Turkey	Other
FRA	0.20	0.00	0.20	0.00	1.00	0.13	0.40	0.50
Early	0.40	0.50	0.60	1.00	0.00	0.71	0.35	0.10
Mid	0.13	0.00	0.00	0.00	0.00	0.08	0.00	0.20
Ref	0.13	0.25	0.20	0.00	0.00	0.08	0.17	0.00
PreSMCRA	0.13	0.25	0.00	0.00	0.00	0.00	0.08	0.20

*Trap nights = 342

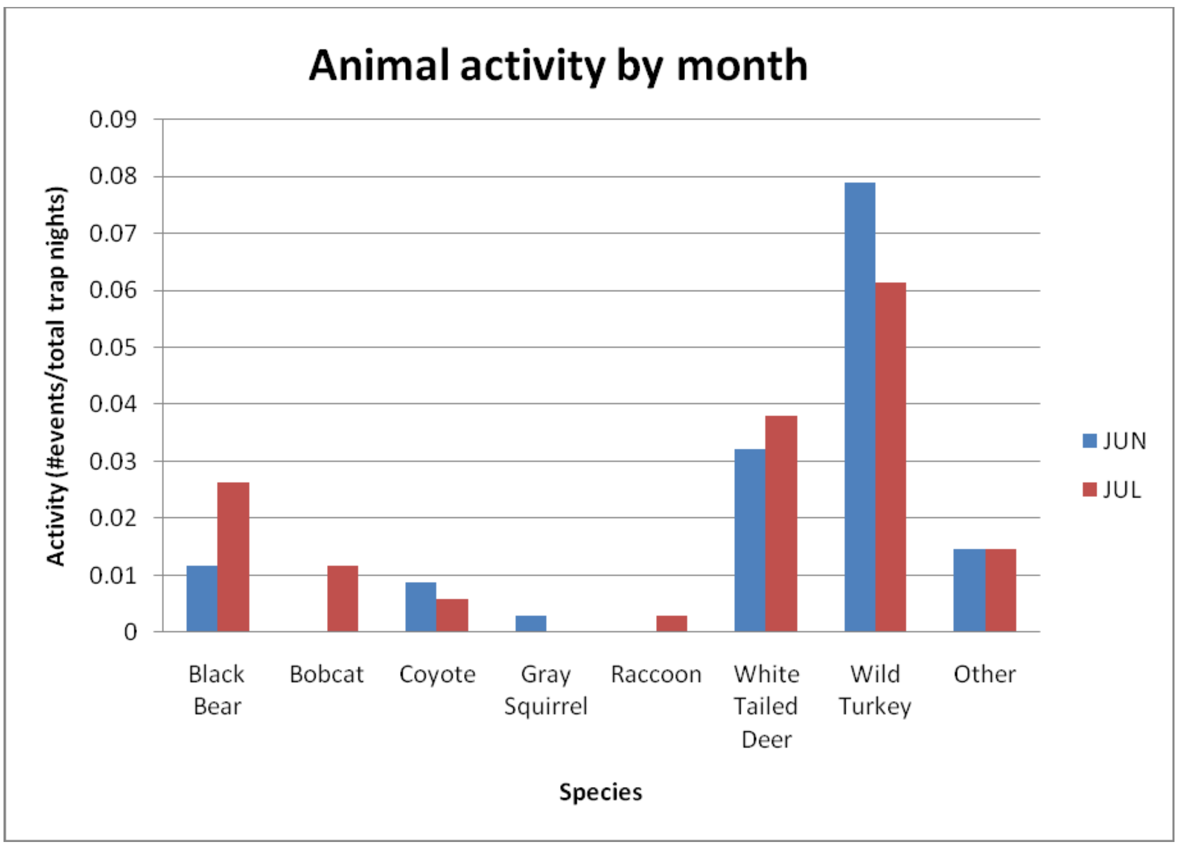

Figure 3. Animal activity by month expressed as # of independent capture events/ trap nights per station.

Arthropod sampling
Two methods were used to sample arthropods between mid-June and mid-July. In pasture cover types, insects were sampled using sweep netting. A 15 cm diameter insect net was used to sample insects along a 20m transect. A total of 4 transects (one in each cardinal direction) were walked 5 times each for a total of 20 sweeps per point. After each pass with the sweep net, insects were placed in bags, labeled and placed in a freezer within 1hr of capture. After 48 hours, insects were identified to order and placed in one of 3 different size classes.
In areas where woody vegetation was present, insects were sampled using a modified version of the branch clipping method described by Johnson (2000). This method involved placing a collapsible bag over a branch and clipping it with shears. Clippings were then shaken vigorously inside the bag to dislodge insects from the vegetation. The vegetation was then removed from the bag and inspected for insects. Once the vegetation was inspected, the observer would place their head inside the bag and identify all insects to order and determine a size classes for each individual. Clippings were taken at 5m intervals along a 25m transect in each cardinal direction from each sampling point, for a total of 20 clippings per point. A total of 2 points were sampled in each of 3 cover types (Table 3).

Table 4. Arthropod sampling method and number of samples for each cover type on the Powell River Project.

Cover Type	# Samples	Sampling Method
FRA	2	Branch Clip
Traditional	2	Branch Clip
Pasture	2	Sweep Net

Expected Outcomes

Ultimately, we will obtain a better understanding of which species use reclaimed mine sites, and which practices used during the reclamation process are most attractive to wildlife. Because of the sensitivity of many wildlife species to disturbance, we hope to use our results to determine the potential of mined sites to support wildlife and suggest reclamation efforts that could be used to attract wildlife. With this knowledge, we will be able to recommend reclamation practices that provide adequate habitat for wildlife and support wildlife habitat as a viable land use under current SMCRA requirements.

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Reclaiming Mined Land for Forests and Forestry

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This report summarizes ongoing research and education activities concerning reforestation of mined lands conducted during the 2009-2010 fiscal year. It is accompanied within this volume by other manuscripts that review more detailed results for certain activities, and by citations of other research and outreach reports. Activities conducted during past year follow below:

Project 1. Grading and Ground Cover for Reclaiming Mined Land for Forestry.

This extensive project was installed in 2008 with the cooperation of Red River Coal Co., Paramount Coal Co., Alpha Natural Resources, Penn-Virginia Resources, and Forest Land Group. It entails testing, the effects of mine-reclamation grading and groundcover seeding effects on hardwood reforestation success. Primary research activities have been conducted by Christopher Fields-Johnson, who is currently preparing his M.S. thesis to include results of this work. We are grateful to the OSM Applied Science Program for providing grant funding to initiate the research, and to the mining firms that prepared experimental areas for research use, Red River Coal Co. and Paramount Coal Co./Alpha Natural Resources, on their coal mining operations. Thanks also to Virginia Department of Mines, Minerals and Energy for their assistance and for accommodating these research areas on active permits.

Project 2. Use of Reclaimed Mined Land for Forest Biomass Production.

This work is being conducted for the purpose of determining the feasibility of using reclaimed mined land to produce woody biomass materials feedstocks for co-firing coal to produce electricity and/or conversion to liquid biofuels and other biobased products. We are indebted to Alpha Natural Resources for providing funding to supplement the Powell River Project general support for reforestation research; the additional funding provided by Alpha was essential to our ability to continue this research. Three separate ongoing projects are contributing to this work:

Tree Species and Density Effects on Woody Biomass Production on Unused Mined Lands: This project was installed in early 2008 on 3 sites owned and managed by Red River Coal Co., Penn Virginia Resource Partners, and Forestland Group, respectively. Second-year results are reported here with a separate manuscript.

Hybrid Poplar Biomass Production Clone Comparison Trials: This project was established in early 2009 with funding from the Virginia Tobacco Commission. Research results from the “Reforestation Techniques for Post-SMCRA Unused Mined Lands” project below demonstrated the potential of hybrid poplar as a biomass producer, and influenced our decision to initiate this work. The Alpha Natural Resources funding for biomass research has been used to support its maintenance and management during 2010. First-year results are reported here with a separate manuscript.

<u>*Stem Form and Fertilizer Effects on Black Locust Biomass Production on Mined Lands*</u>: Early results from the "Tree Species and Density Effects" trial reported above demonstrate that black locust produces large amounts of biomass during its first and second year; and observations reveal its continued rapid growth. A problem with using black locust for biomass production on coal surface mines is its poor stem form. This trial was installed with black locust stock procured and donated by Williams Forestry and Associates. The study is comparing the performance of black locust varieties that have been selected for their superior growth form to stock procured from sources that commonly provide for coal surface mines. The study was installed in early 2010; its purpose and design are described in this volume with a separate manuscript.

Project 3. Which Topsoil Substitutes are Best for Growing Trees?

The objective of this study is to determine how mine soils weather with time and how they may become more suitable for trees. The study is being conducted on the Controlled Overburden Placement (COP) experimental site, which was established in 1982 and is the longest intact and continuously monitored experimental manipulation of topsoil substitutes and organic amendments in the world. From 1983 to 2002, the forested side of the plots were predominately under pitch x loblolly pine hybrid vegetation. In 2002 the pines were harvested, providing valuable information on the influence of topsoil substitutes and organic amendments on the potential productivity of reforestation efforts. Following this rotation, and with an increasing interest in the return of native hardwood species to the post mining landscape, the COP plots were planted with northern red oak seedlings. In October 2009, following their eighth growing season, all 180 (9 trees x 5 treatments x 4 replicates) red oak were harvested to similarly evaluate their productivity in response to the different treatments. A new, incoming graduate student will combine this information with a focus on soils in order to better understand how the properties and process of the post-mining landscape that can help facilitate the vision of the Forestry Reclamation Approach (FRA).

Project 4. Reforestation Techniques for Post-SMCRA Unused Mined Lands.

Research is being conducted to determine silvicultural methods and economic benefits of converting grasslands to productive forests. This project was initiated with US Department of Energy funding in 2004. Half-acre plots of three planted forest types (hybrid poplar, white pine, and mixed hardwoods) were each treated with three levels of management (weed control, weed control + tillage, and weed control + tillage + fertilizer). Three of the nine replications of this experiment are located on or near the PRP. Hybrid poplar growth on these plots far exceeds other species and influenced our decision to become engaged with the hybrid poplar clonal trials study, as described above. Christopher Fields-Johnson is reporting 5-year results for this study in his M.S. thesis. In addition, we are working with The Nature Conservancy to aid their reforestation of a former Pittston Coal Co, site at Hazel Mountain, Virginia, near the border of Wise, Dickenson, and Russell Counties.

Project 5. American Chestnut Restoration.

Hybrids of chestnut that are botanically indistinguishable from American chestnut (*Castanea dentata*) and have the blight-resistance of Chinese chestnut (*Castanea mollissima*) are being developed by the American Chestnut Foundation. Reclamation of mined land in the Appalachians can aid the introduction of these hybrids because of the coincidence of the Appalachian coalfield with the central range of the American chestnut and because of the large areas of land opened up by mining that are available for afforestation. We have two ongoing field trials in place that test the effect of mine reclamation and planting methods on chestnut re-establishment success. Those results are reported here in a separate manuscript.

Project 6. Education Programs and Demonstration Forests.

During the past year, one of our group served as the co-chair of the 4th annual Appalachian Regional Reforestation Initiative Conference in Pittsburgh, PA, (June 2010); and we led a field program for Virginia Department of Mines, Minerals and Energy and US Office of Surface Mining personnel at Powell River Project Research and Education Center in September 2009. We also completed several outreach publications, as listed below.

Publications:

J. Showalter, J. Burger, C. Zipper. 2010. Hardwood seedling growth on different mine spoil types with and without topsoil amendment. Journal of Environmental Quality 39: 483-491.

J.A. Burger, D.M. Evans. 2010. Ripping compacted mine soils improved tree growth 18 years after planting. p. 55 - 69, in: Proceedings, National Meeting of the American Society of Mining and Reclamation.

J.A. Burger, C.E. Zipper. 2010. Reforestation Guidelines for Unused Surface Mined Lands in the Eastern United States. Virginia Cooperative Extension Publication 460-144.

V. Davis, J. Franklin, C. Zipper, P. Angel. 2010. Planting Hardwood Tree Seedlings on Reclaimed Mine Land in Appalachia. Appalachian Regional Reforestation Initiative, Forest Reclamation Advisory Number 7.

D.M. Evans, C.E Zipper, J.A. Burger, C. Fields-Johnson. 2010. Tree species and density effects on woody biomass production on unused mined lands: Establishment and two-year results. p. 276 -291, in: Proceedings, National Meeting of the American Society of Mining and Reclamation.

C. Fields-Johnson, J. Burger, D. Evans, C. Zipper. 2010. Response of improved American chestnuts to planting practices on reclaimed surface mined land. p. 319 -336, in: Proceedings, National Meeting of the American Society of Mining and Reclamation.

C. Fields-Johnson, C. E. Zipper, J.A. Burger and D.M. Evans. 2010. Second year response of Appalachian mixed hardwoods to soil surface grading and herbaceous ground cover on reclaimed mine land. p. 305 -318, in: Proceedings, National Meeting of the American Society of Mining and Reclamation.

Second Year Response of Appalachian Mixed Hardwoods to Soil Surface Grading and Herbaceous Ground Cover

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Abstract. Recent experience suggests that native Appalachian hardwood trees can be successfully established on coal surface mining sites if appropriate reclamation techniques are used. The Forestry Reclamation Approach (FRA) is a set of mine reclamation techniques developed for that purpose. Questions remain regarding how soil surface grading and choice of herbaceous vegetation affect tree survival, soil erosion and plant succession. An experiment was begun in the spring of 2008 with the goal of evaluating effects of grading and hydroseeding treatments prescribed by the FRA on reforestation success. Three steep (approximately 60% slopes) reclaimed mine sites were prepared in the coalfield of southwest Virginia. Half of each site was graded using conventional pre-FRA grading practices that cause compaction of the surface soil. The other half was loose-graded as per FRA recommendations. Within each grading treatment at each site, one third of the area was seeded with a conventional herbaceous vegetation mix that included competitive grasses and legumes; one third with a tree-compatible herbaceous mix comprised of less competitive grasses and legumes; and one third with 100% annual ryegrass. All experimental areas were planted with the same mix of native hardwood trees. Tree survival over one growing season was similar on the loose (71%) and compacted (70%) grading treatments, as well as on the conventional (65%), tree-compatible (71%), and annual ryegrass (75%) revegetation treatments. Recruitment diversity of non-planted species was greatest on the annual ryegrass treatment (12 volunteer species), suggesting this re-vegetation practice creates the most favorable conditions for natural succession. Soil erosion rates were significantly higher on the compacted treatment (-8mm soil soil surface change) than on the loose-graded treatment (+10mm soil soil surface change) over the course of two years. The annual ryegrass treatment produced significantly less ground cover (55% total) after two years than the conventional ground cover treatment (83% total), but this did not result in greater soil erosion.

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[1] This paper has been previously published as: C. Fields-Johnson, C. E. Zipper, J.A. Burger and D.M. Evans. 2010. Second year response of Appalachian mixed hardwoods to soil surface grading and herbaceous ground cover on reclaimed mine land. p. 305 -318, in: Proceedings, National Meeting of the American Society of Mining and Reclamation. .ASMR, 3134 Montavesta Rd., Lexington, KY 40502.

Introduction

Recent progress has been made in the science and implementation of the Forestry Reclamation Approach (FRA), guidelines used for revegetating lands disturbed by surface mining for coal in the Appalachian region. The FRA is a mine reclamation protocol designed to improve the establishment of high-value hardwoods, increase the survival and growth of planted trees, and accelerate forest succession. The FRA has been approved by regulatory agencies and can be implemented more cost-effectively than traditional mine reforestation approaches prescribing extensive soil grading and dense herbaceous cover (Burger and Zipper, 2002). The FRA is intended to restore forested ecosystems on reclaimed mine sites and to recognize that productive forests produce economically valued products such as harvestable timber while providing ecosystem services such as production of clean water and air, sequestration of atmospheric carbon and provision of wildlife habitat (Angel et al., 2005).

Key aspects of the FRA include loose grading of the soil surface and tree-compatible ground covers. Low compaction grading helps planters install trees at the proper depth, allows rain to readily infiltrate the soil rather than causing erosive surface flow, increases soil moisture availability, improves soil aeration, and facilitates root growth by the planted trees. Low compaction grading is less expensive than conventional grading practices because fewer passes with grading equipment are required (Sweigard et al., 2007). Tree survival and growth are generally higher on loose-graded mine sites than on compacted and tracked-in mine sites. Numerous studies have demonstrated that high soil bulk density, which occurs as a result of excessive soil compaction, has a negative effect on tree growth (Jones et al, 2005; Rodrigue and Burger, 2004; Andrews et al, 1998; Torbert and Burger, 2000; Torbert and Burger, 1990).

Low density, low-growing herbaceous vegetation minimizes soil moisture competition and allows sufficient light penetration for tree seedling growth. Because of the vigorous nature of many forage species used for hay or pasture applications, most are not conducive to tree seedling establishment and growth. These species include Kentucky-31 tall fescue (*Festuca arundinacea*), red clover (*Trifolium pratense*) and sweet clover (*Melilotus alba*). Less-competitive legumes, considered to be more compatible with tree survival and growth and commonly recommended for use in the FRA, include birdsfoot trefoil (*Lotus corniculatus*) and white or ladino clover (*Trifolium repens*), while recommended annual grasses include foxtail millet (*Setaria italica*) and annual ryegrass (*Lolium multiflorum* Lam.). Perennial grasses that are considered "tree compatible" include perennial ryegrass (*Lolium perenne*), timothy *(Phleum pratense)* and orchardgrass (*Dactylis glomerata*) on steep slopes. Weeping lovegrass (*Eragrostis curvla*) is a tall grass that is useful on low soil pH sites at low seeding rates (Burger and Zipper, 2002).

Goal and Objectives

The goal of this study was to assess the effects of different surface grading techniques and herbaceous ground covers on forest ecosystem re-establishment on active mining operations.

We tested the following hypotheses:

- Increased intensity of grading and tracking by mining equipment reduces the survival of planted native hardwood trees and accelerates soil erosion.
- Increased levels of competitive herbaceous ground cover reduce the survival of planted native hardwood trees and hinder the recruitment of native vegetation.

Testing these hypotheses will allow refinement and improvement of the FRA prescriptions with a corresponding improvement in survival of planted trees and accelerated forest succession. This paper reports second year results for a study that was also described, after the first year, by Fields-Johnson et al. (2009).

Methods and Materials

Overview of Treatments and Design

Three experimental sites (blocks) were established by cooperating mining firms on active mining sites in southwestern Virginia. At each site, two grading treatments and three ground cover vegetation treatments were installed as a 2 x 3 factorial randomized block design, resulting in 6 treatment combinations and 18 total treatment plots. Each block was approximately 2.5 ha and the treatment plots averaged approximately 0.4 ha in size, although individual treatment plot sizes varied from this average. The two grading treatments were 1) smooth-grading with tracking-in (i.e. covering the surface with dozer cleat marks) or back-blading (dragging the bulldozer blade backwards across the site to create a smooth surface); and 2) loose-grading with a single dozer pass. Three revegetation treatments were sown on each grading treatment plot: 1) a conventional mix of herbaceous species intended to create >90% ground cover within the first few months of a growing season after seeding, 2) a tree-compatible mix (designated as "Powell River Project mix" in Fields-Johnson et al., 2009) intended to create a moderate level of initial ground cover while eventually fully covering the soil surfaces, and 3) annual ryegrass, intended to create the lowest level of groundcover by planted species (Table 1).

The conventional ground cover treatment seed mix prescription is one that is commonly applied by a commercial hydroseeding firm on coal mining operations in southwestern Virginia. The tree-compatible mix prescription has been developed by the researchers using a process of trial, error and observation of many herbaceous species over many years. Hydroseeding was performed by a commercial contractor using operational procedures, under supervision by the mining firms but using our prescriptions, following final grading of mine spoil. Fertilizer was prescribed for inclusion in all hydroseeding mixtures at an approximate rate of 22 kg ha^{-1} nitrogen (N), 68 kg ha^{-1} phosphorous (P) and 18 kg ha^{-1} potassium. This fertilization prescription for reforestation has been developed by trial and error as a way to provide trees ample P without causing excessive herbaceous growth with large amounts of N. Block 1 was hydroseeded in the fall of 2007, Block 2 was hydroseeded in the winter of 2007-2008, and Block 3 was hydroseeded in early spring of 2008. Mining was completed for these sites at different times, hence the staggered hydroseeding schedule.

All sites were planted with the same mix of native trees (Table 2) by a commercial tree-planting contractor in mid-January of 2008. The tree species mix prescription has also been developed by trial, error and observation and included 205 trees ha^{-1} for each of seven commercially valuable hardwoods, lesser rates for two other commercial species, and low rates for several species of specific wildlife value. The planting contractors modified the actual planting rates based on available nursery stock and deviated somewhat from the planting prescription. These trees were all planted as one-year-old, bare-root seedlings without supplemental watering or fertilization. The overall tree survival rate in 2008 was 39% (Fields-Johnson et al. 2009), a rate considered unacceptably low by reclamation standards. As a result, all sites were re-planted in January of 2009 to bring them back to full stocking (Table 2). Photographs and maps for treatments and block locations can be found in Fields-Johnson et al. (2009).

Erosion Measurement

Erosion pins made of 1/2-inch (1.25 cm) diameter steel rebar were used to estimate loss and accumulation of surface soil. Nine erosion pins were driven into the ground to a depth of approximately 60 cm in each of the 18 treatment plots. Once installed, the sections of the pins that remained exposed were measured in height to the nearest mm on the uphill side. Thereafter, the pins were measured before the growing season (May) and after the growing season (November) of each year.

Table 1. Prescribed seed and mulch mixtures for ground cover treatments.

Annual Ryegrass Only	Rate
Seed Mix:	(kg ha^{-1})
Annual ryegrass (*Lolium multiflorum*)	22
Wood Cellulose Fiber	1680
Tree-Compatible Mix	Rate
Seed Mix:	(kg ha^{-1})
Annual ryegrass (*Lolium multiflorum*)	22
Perennial ryegrass (*Lolium perenne*)	11
Timothy (*Phleum pretense*)	6
Birdsfoot trefoil (*Lotus corniculatus*)	6
Ladino clover (*Trifolium repens*)	3
Weeping Lovegrass (*Eragrostis curvula*)	2
Wood Cellulose Fiber	1680
Conventional Mix	Rate
Seed Mix:	(kg ha^{-1})
Rye grain (*Secale cereale*)	34
Orchardgrass (*Dactylis glomerata*)	22
Perennial ryegrass (*Lolium perenne*)	11
Korean lespedeza (*Lespedeza cuneata*)	6
Birdsfoot trefoil (*Lotus corniculatus)*	6
Ladino clover (*Trifolium repens*)	6
Redtop (*Agrostis gigantea)*	3
Weeping lovegrass (*Eragrostis curvula*)	2
Wood Cellulose Fiber	1680

Soil Sampling and Testing

Soil samples were gathered for each of the 18 plots in the Spring of 2008. Samples were composed of nine sub-samples taken within each plot, each taken one meter from an erosion pin, and composited. The surface 5cm of soil were removed in order to eliminate hydroseeding materials from the sample and a 10-cm depth sample taken (i.e. 5 – 15 cm below the soil surface). Soil samples were air dried then sieved through a #10 screen to separate coarse and fine fractions. Fines were analyzed for pH, extractable cations, cation exchange capacity, soluble salts and organic carbon content (Soils data in Fields-Johnson, 2009). The coarse fragment fraction (fragments >2mm) was analyzed to determine the percent of each major rock type (weathered brown sandstone, un-weathered gray sandstone, siltstone, black shale, and coal).

Table 2. 2008 planting prescription and actual survival rates and 2009 re-planting prescription for trees to be planted alongside surviving trees to replace trees lost to mortality.

	2008 Planting Prescription	**2008 Survival**		**2009 Re-planting Prescription**
Species	**(trees ha^{-1})**	**(trees ha^{-1})**	**Rate[a]**	**(trees ha^{-1})**
White Ash (*Fraxinus americana*)	205	138	67%	67
White Oak (*Quercus alba*)	205	119	58%	86
Redbud (*Cercis canadensis*)	54	27	50%	27
Gray Dogwood (*Cornus racemosa*)	54	27	50%	27
Red Mulberry (*Morus rubra*)	25	12	49%	13
Black Cherry (*Prunus serotina*)	205	93	45%	112
Red Oak (*Quercus rubra*)	205	88	43%	117
Chestnut Oak (*Quercus prinus*)	205	67	33%	138
Black Oak (*Qurecus velutina*)	205	65	32%	140
Yellow-poplar (*Liriodendron tulipifera*)	124	33	27%	91
Sugar Maple (*Acer saccharum*)	205	28	14%	177
White Pine (*Pinus strobus*)	91	6	6%	85
Shagbark Hickory (*Carya ovata*)	62	3	4%	59
Total	**1,845**	**728**	**39%**	**1,139**

[a] Calculated from prescribed planting rate, which may have differed from the actual rate.

Vegetation Sampling

Five 0.02-ha, circular, woody-plant measurement plots were established on each treatment plot. Species, ground-line diameter, and distance from soil surface to highest live bud were measured for all trees within measurement plots in November of 2009. Additionally, four 0.0004 ha circular herbaceous plant plots were nested inside of each woody plant measurement plot. Within each measurement plot an ocular estimate of total living and dead ground cover percent was made in August of 2009 by comparing observed coverage with pre-established diagrams of various coverage rates. Samples of all observed plant species were collected for identification and separated into "planted" versus "volunteered" categories in order to distinguish the origin of each species.

Statistical Analysis

Data were analyzed using JMP 7.0 (SAS Institute Inc., Cary NC). Differences among treatments were determined using a randomized block ANOVA. Tukey-Kramer HSD was used for mean separations ($P < 0.10$). Multi-factor analysis was also performed to analyze treatment interactions and block effects.

Results

Significant differences were found in rock composition in the coarse fraction of the loose versus the compacted treatments (Table 3). Compacted treatments had higher average levels of weathered sandstone spoil, whereas loose treatments had higher levels of un-weathered sandstone. Amount of compaction had no significant effect on herbaceous ground cover or tree survival over the course of the 2009 growing season (Table 4). One mining firm reported that it required approximately 7.5 to 8.5 additional machine hours per ha to complete conventional grading treatments compared to loose graded treatments. The conventional ground cover treatment produced significantly more cover than the annual ryegrass treatment, but tree survival differences between ground cover types was minimal. In some cases the exposed height of erosion pins decreased over time, indicating a positive soil surface change (Table 5). This unexpected result was attributed to soil expansion caused by physical unloading, freeze-thaw processes, mineral slaking, moisture swell, and rooting expansion. Hence, erosion-pin measurements are expressed as "surface change," a relative measurement calculated from the pins' exposed heights; with negative change (erosion) indicating increased erosion-pin exposure. Visual observations indicated that soil was being lost even at sites where measured surface change was positive. Loose grading resulted in significantly less apparent erosion (as indicated by measured surface change) than compact grading. The tree compatible and annual ryegrass ground cover treatments eroded nominally less than the conventional mix. Grading treatment had no significant effect on volunteer herbaceous species richness, but the annual ryegrass revegetation treatment allowed more volunteer species to establish than the other two ground cover treatments (Table 5). No significant interaction effects between ground cover type and grading type were found for tree survival and soil erosion rates.

Table 3. Coarse fragment rock type analysis: Percentage by weight of soil samples made up of > 2mm coarse fragments, and percentage by volume of > 2mm fragments made up of weathered brown sandstone, un-weathered gray sandstone (with significant difference in means by Tukey HSD between grading treatments indicated by different letters beside values, α = 0.10), siltstone, shale and coal.

Treatment	Coarse Fragments	Weathered Sandstone	Un-weathered Sandstone	Siltstone	Shale	Coal
Grading:						
Compact	59%	47% a	15% b	36%	1.1%	1.4%
Loose	59%	39% b	28% a	31%	0.7%	1.6%
Groundcover:						
Conventional Mix	59%	46%	23%	29%	0.3%	0.8%
Tree Compatible Mix	58%	45%	18%	33%	1.3%	2.5%
Annual Ryegrass Only	62%	38%	22%	38%	1.0%	1.3%

Table 4. Treatment effects on percent ground cover rates and surviving trees per acre with significant differences (Tukey HSD) by alpha (α) level with differences indicated by different letters beside values within categories.

Grading	Ground Cover	α = 0.10	Tree Survival Rate	α = 0.10
Compact	72%	a	70%	a
Loose	70%	a	71%	a
Ground Cover				
Conventional Mix	83%	a	65%	a
Tree Compatible Mix	75%	ab	71%	a
Annual Ryegrass Only	55%	b	75%	a

Table 5. Cumulative treatment effects on surface change over the 2008 and 2009 growing seasons and on number of 2009 volunteer species. Significant differences (Tukey HSD) are depicted by different letters beside values within categories.

Grading	Surface Change (mm)	α = 0.10	Volunteer Spp.	α = 0.10
Loose	10	a	8	a
Compact	-8	b	6	a
Ground Cover				
Annual Ryegrass Only	8	a	12	a
Tree Compatible Mix	2	a	5	b
Conventional Mix	-7	a	4	b

Discussion

Neither the amount of ground cover nor tree survival was significantly affected by grading treatments after the second growing season (Table 4), a result similar to that found by Torbert and Burger (1992). The confounding factor of spoil selection may have biased the effects of grading treatments (Table 3). Past experiments have shown that tree survival and growth for most species improves with reduced grading activity (Angel et al., 2006). Loose graded plots in this experiment had significantly more un-weathered sandstone ($p = 0.01$) and less weathered sandstone ($p = 0.10$) than compacted plots. Research demonstrates that weathered sandstone in this region is an excellent substrate for growing native trees (Emerson et al., 2009; Showalter and Burger, 2006), and weathered sandstone materials are recommended for surface placement where available (Burger et al., 2005). Although efforts were made to control spoil selection during experimental plot construction, variability in local sources resulted in measurable differences in spoil type between grading treatments. Another possible explanation for grading not having an effect on tree survival may relate to plot steepness (often 60% slopes or steeper). Mining equipment creates less compaction on steep slopes than on near-level grades. Although we did not find a difference between the grading treatments at year two, the long term response on very steep slopes has yet to be determined. Other research has suggested that soil compaction effects may be more evident in the long-term (Burger and Evans, 2010).

Although significant differences in ground cover rates were achieved by the different herbaceous vegetation treatments, reduced plant cover did not result in significantly increased tree survival. Nominally, tree survival did vary inversely with ground cover percentage across the three treatments (Table 4), indicating that significant relationships might emerge with more time. In the first year after replanting, tree survival was considered acceptable across the entire experiment at 65-75%. The positive effects of lower herbaceous planting rates on tree survival on reclaimed mined land have been demonstrated (Torbert et al., 2000, Burger et al., 2005b, Skousen et al., 2006). Tree-compatible and annual ryegrass only ground cover treatments may produce significant long-term differences in tree growth and survival due to the reduced competition between trees and herbaceous vegetation for water, sunlight and soil nutrients as the transplanted trees move from the establishment to the growth phase. Re-planting brought all plots up to full stocking before the 2009 growing season, during which rainfall was abundant. The fact that re-planted trees were not subjected to significant moisture stress during their first summer may have influenced the lack of observed effect by grading treatment on tree survival.

We hypothesized that higher levels of compaction would lead to higher levels of surface erosion, and this basic hypothesis is supported by our study results (Table 5) and those of Torbert and Burger (1992). Our hypothesis was based on previous research findings that the increased erosion associated with greater grading intensities results from reduced soil macro-porosity and poor water infiltration (Evans and Loch, 1998) caused by excessive grading, but the exact mechanism causing our results has not been determined. Although our results did not demonstrate an effect by grading on tree survival, other effects, such as reduced grading costs and decreased soil erosion, are also important reasons for preferring minimization of surface grading to limit soil-surface compaction.

Our hypothesis that grading treatments would exert primary controls over erosion rates, suggests that all three experimental ground cover treatments would control erosion equally well. This was the case at the end of year two (Table 5). Past study has shown that ground cover with as little as 50% coverage can drastically reduce runoff and erosion compared to bare soil (Loch, 2000). Even though the annual ryegrass ground cover died back after the first year, this treatment resulted in the least nominal soil erosion. Heavy first-year growth of annual ryegrass created a dense mat of dead biomass that protected the site the second year. Furthermore, the ryegrass cover allowed more recruitment of non-planted species (Figure 1), which may lead to additional reductions in erosion over longer intervals.

We hypothesized that lower cover rates would facilitate faster recruitment of volunteer plants and succession relative to the other two ground cover treatments. Earlier studies have shown that plantings of non-native, aggressive ground covers can impede herbaceous plant succession (Holl, 2002; Burger et al. 2005b). The annual ryegrass treatment did have a significantly higher number of volunteer herbaceous species at the end of the second growing season (Table 5), most likely the result of reduced cover by living plants and reduced herbaceous competition. Contaminants in the annual rye seed may have also contributed to these other observed species, although we consider that to be unlikely because several of the plant species observed did not appear on the other ground cover treatment sites, where annual ryegrass was also a seed mix component, and because the seed sources were certified to have low levels of contamination. Planting only annual ryegrass may speed succession without increasing erosion or reducing tree survival. Annual ryegrass treatments have been shown to be compatible with tree establishment while achieving other reclamation goals (Groninger et al., 2007), especially at locations with near native seed sources and with soil properties that are favorable for native volunteer species establishment.

One concern with seeding practices that produce low levels of groundcover, such as the annual ryegrass in this experiment, is that they may result in exotic species invasion along with desirable natives. An alternative hypothesis is that less competitive groundcovers result in faster native plant recruitment, thus reducing the potential for exotic species invasion (Burger et al., 2009). Past study has shown that native trees can become established on sites even when they are not planted, where aggressive groundcover is not present (Skousen et al., 2006). Our finding that seeding with annual ryegrass is compatible with native plant recruitment is consistent with this earlier study. Further monitoring will be necessary to determine how herbaceous ground covers affect long-term recruitment of both desirable and un-desirable species. Another longer-term question regarding the annual ryegrass treatment, which can only be answered with long-term monitoring, is whether the lack of planted N-fixing legumes will negatively affect available soil N and, as a result, decrease forest productivity.

Figure 1. Research personnel on the loose grading, annual ryegrass treatment of Block 1 in late summer of experimental Year 2. The photo shows how a variety of unplanted species are being recruited to the reclaimed area.

Conclusions

After two growing seasons no significant differences in survival rates of mixed native hardwoods were observed to occur in association with, or caused by, different grading and ground cover treatments, but rates of soil erosion and unplanted species recruitment were affected. On the steep slopes of these experimental sites, loose-graded reclamation treatments had less soil surface loss, an indicator of soil erosion, than the more intensively graded treatments. Considering the fewer machine hours required for loose grading, these study results support the loose-grading methods recommended with the FRA (Sweigard et al., 2007). Planting only annual ryegrass appears to be an appropriate technique for promoting faster ecological succession on reclaimed mined lands on these study sites, without any associated negative effects such as increased rates of soil erosion. How widely or generally this finding can be applied on coal mine reclamation sites has yet to be determined.

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Tree Species and Density Effects on Woody Biomass Production on Unused Mined Lands[1]

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Abstract. Under-utilized, previously mined lands may be used to produce woody biomass materials for energy production and C sequestration. Past research trials have shown that tree growth on mined lands can be highly productive if suitable reclamation practices are used. This study tests the productivity of woody biomass plantations on previously mined lands after ripping to reduce soil compaction, using four species treatments under two planting densities. This paper summarizes the establishment procedure and initial results after two years of growth. Initial results indicate that black locust has the highest mean per tree volume growth under both high (1311 cm^3) and low (1917 cm^3) density planting. Due to good survival and high wood density, it also has the highest dry biomass production by an order of magnitude over other species for both high (2.39 Mg ha^{-1}) and low (1.28 Mg ha^{-1}) density planting. Hybrid poplar and American sycamore are secondary in both volume and biomass growth. Red oak and eastern cottonwood had poor survival, volume growth, and biomass production. At year two, planting density did not have a significant effect on volume growth, likely due to lack of crown/root closure between trees.

Introduction

The extensive hardwood forests of the Appalachian Mountains may help meet current and future demand for biomass materials used for energy production. If demand for energy increases or future government policies limit net carbon emissions from energy production, demand for carbon-neutral fuels such as woody biomass products may increase. Carbon sequestration in the United States, using forest practices, may also become an exchangeable commodity under future C regulation scenarios that may encourage production of biomass materials. Localized demand for woody biomass materials may increase if construction of 'hybrid' power plants in the Appalachian region increases. These plants are contracted to burn a percentage of non-coal materials, which will increase demand for biomass products in former and current coal producing areas. For example, Dominion Resources' Virginia City Hybrid Energy Center, currently under construction in Wise County, Virginia will use up to 20 percent biomass for its fuel (Dominion Resources 2009). Hybrid power plants that include biomass materials in their fuel mix, will place pressure on local forest systems to produce acceptable forest products. If they prove capable of producing biomass fuels in an economically viable fashion and in sufficient quantities, under-utilized mined lands may help to provide these needed biomass materials in the coal fields of Virginia and surrounding coal producing states.

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There are many advantages and benefits of using mined lands to produce biomass materials. The proximity of mined lands to power plants built in the coalfields will enable provision of biomass products to such plants, while limiting or minimizing transportation costs. If such mined lands can be converted and managed to provide marketable biomass products, it may reduce the impact on native forests, which can be conserved for traditional forest products and ecosystem services. Local economies will also benefit through the production of a renewable forest product on lands deemed unproductive by past generations and largely ignored by the forest products industry.

Research trials have demonstrated that properly reclaimed mine-lands can be highly productive when properly reclaimed and planted with trees (Amichev et al. 2008, Burger and Fannon 2009, Fields-Johnson et al. 2008, Amichev et al. 2004, Burger 2004). Mine soils can offer soil-like materials comprised of freshly fractured rocks at thicknesses far deeper than many of the region's natural soils. These freshly-fractured geologic materials often have chemical characteristics, that are favorable to plant growth, including pH levels and nutrient cation availabilities (Anderson et al. 1989). Research on native soils in mountainous areas suggests that productivity (Mg ha^{-1} yr^{-1}) of fast growing woody crops growing on favorable sites, can be 3 to 5 times greater than long-rotation natural forests (Amichev 2007). For example, in an analysis of data collected from eight un-mined native hardwood forests adjacent to coal mining areas, Amichev (2007) found that total-tree carbon accumulation averaged 2.1 Mg C ha^{-1} yr^{-1} over 60 year rotations. In contrast, he estimated that hybrid poplar growing on favorable sites with short rotations in a similar climate have the potential to accumulate total-tree C at a rate of 11 Mg C ha^{-1} yr^{-1}. Amichev et al. (2004; 2008) also found that pre-mining C sequestration can be restored on many low-quality forested sites, indicating that these mined lands can be used to produce woody biomass at levels similar to or above those of low-quality native sites. Rodrigue et al. (2002) studied forest growth on older coal mine sites in the eastern and Midwestern United States that were known to support forests, and found that 12 of 14 sites achieved productivities similar to nearby un-mined forests. The capacity of mined lands to produce woody biomass, when prepared and managed specifically for that purpose, has not been explored.

More than 40,000 ha in the southwestern Virginia coalfield, and about 0.7 million ha throughout Appalachia, have been mined for coal and reclaimed under the Surface Mining Control and Reclamation Act of 1977. Many of these areas remain accessible because mining access roads were left in place. However, past mining practices have left many sites with highly degraded site productivity due to soil compaction and lack of tree compatible vegetation. We hypothesize that these degraded mined sites with favorable soil chemical characteristics can be managed for intensive tree biomass production regardless of past soil and vegetation management. Concurrent with the production of biomass materials, we also hypothesize that stands of fast growing trees on formerly mined lands may be a viable C sequestration technique. This study is designed to test the capability of previously mined lands to be used for tree biomass production. It tests biomass production for differing tree species and planting densities on previously mined lands that have been ripped to improve soil physical properties. We used an adaptation of the Forestry Reclamation Approach (FRA), a reclamation method developed for preparing active mine sites to support woody vegetation using forestry techniques (Burger et al. 2005), to prepare these older mine sites. This paper summarizes our methods for planting and maintenance of 6 ha of operational biomass plantation at three sites. It also summarizes initial results from two years of growth, with a focus on production of aboveground biomass. There are four primary objectives for this study.

1. Develop and describe a method for preparing mine sites that have been reclaimed in previous years and are currently unused for biofuels production.
2. Measure and compare production of woody biofuel crops on mined lands using various species and planting densities.
3. Measure and compare optimum harvest cycles of woody crops on mined land.
4. Determine the potential of woody biomass, growing under optimal soil conditions, to sequester atmospheric carbon in above and below-ground forms.

This paper presents the results for the first two objectives. Objectives 3 and 4 will be addressed at future measurement intervals.

Methods

Three sites in Wise County, Virginia were included as replicate blocks (Figure 1). This mountainous area of Virginia receives approximately 120 cm of mean annual precipitation and has a mean annual temperature of 12°C. The native vegetation types on these mountains are dominated by diverse, mixed hardwood forests. Previous to the study installation, vegetation at all three blocks was unmanaged grasses and woody shrubs, and had no significant issues with salts or metalloids. The sites were not used for agricultural grazing, but did include remnants of the original reclamation tree plantings. Block 1, the Red River site at 806 m in elevation, was reclaimed in the early 2000s with typical reclamation grasses and *Pinus* sp. Remnant pine survival was poor, as pines occupied ~10% of the site, and herbaceous vegetation was sparse. Block 2, the Across The Road site at 686 m, was reclaimed in the mid-1980s. Vegetation was a dense mixture of early successional volunteer species. Block 3, The Bean Gap site at 616 m, was dominated by grasses, with sparse trees that had survived the initial reforestation in the late 1990s. The original reclamation was with a mixture of native hardwoods and eastern white pine. Survival was poor, likely due to physical compaction effects caused by equipment operation on this relatively flat to gently sloping area.

Each site provided 2 ha of relatively flat ground (<15% slope), without large established woody vegetation, that can be reached by heavy equipment for site preparation and harvesting. In December, 2007, each site was disked and ripped to till under existing vegetation and to alleviate possible compaction, leaving loose soil material for tree planting and root growth. This was accomplished with a heavy forestland disc harrow used to break up the soil, followed by a second pass to deep till and mound the tree planting row. The tillage tool had a 1 meter center shank that ripped a deep trench through the compacted mine soil while large disks around the shank produced a mound of loose soil over the rip where the trees were planted. Smaller shanks to the right and left of the center shank broke up the surface to 1 foot on either side of the planting location.

Each of the three blocks was divided into 4 species treatment areas of approximately 0.5 ha (Figure 2). Species treatment plots were planted with hybrid poplar cuttings (*Populus trichocarpa* L. (Torr. and Gray ex Hook.) x *Populus deltoides* (Bartr. Ex Marsh.) hybrid 52-225), American sycamore (*Platanus occidentalis*), and black locust (*Robinia pseudoacacia*),each at two planting densities. The low density treatment was planted along the 11 foot furrows with an intended target of 3.4 m by 3.4 m spacing or 860 trees ha^{-1} (Figure 3). The high density treatment was planted at half the distance between trees both on the furrows and in-between the furrows with an intended target of 1.7 m by 1.7 m spacing or 3400 trees ha^{-1}. A fourth species treatment included an additional low density treatment of northern red oak (*Quercus rubra*) (3.4 m by 3.4 m) interplanted with rows of eastern cottonwood (*Populus deltoides*) (1.7 m within row). This treatment was included to test the fast growing eastern cottonwood's ability to train the slower growing but higher value red oak. Red oak is a high-value sawtimber species that is native to Appalachian forests. The value of red oak as sawtimber may be increased by training its stem form at an early age by interplanting with eastern cottonwood, which can be harvested for biomass products at a relatively short rotation age. A low density red oak treatment (3.4 m by 3.4 m) without cottonwood was included to compare against the interplanted red oaks. A final treatment of low density mixed hardwoods was included, where space allowed, using a planting mix of: *Prunus serotina*, *Quercus* sp, *Acer saccharum*, *Platanus occidentalis*, *Robinia pseudoacacia*, *Fraxinus* sp., and *Cornus* sp. The hybrid poplar cuttings were purchased from a grower in Oregon, while all other trees were planted by a planting contractor, as seedlings, obtained by the contractor from commercial sources. At the time of planting, trees received no fertilizer, tree protectors, mycorrhizal treatments, or watering.

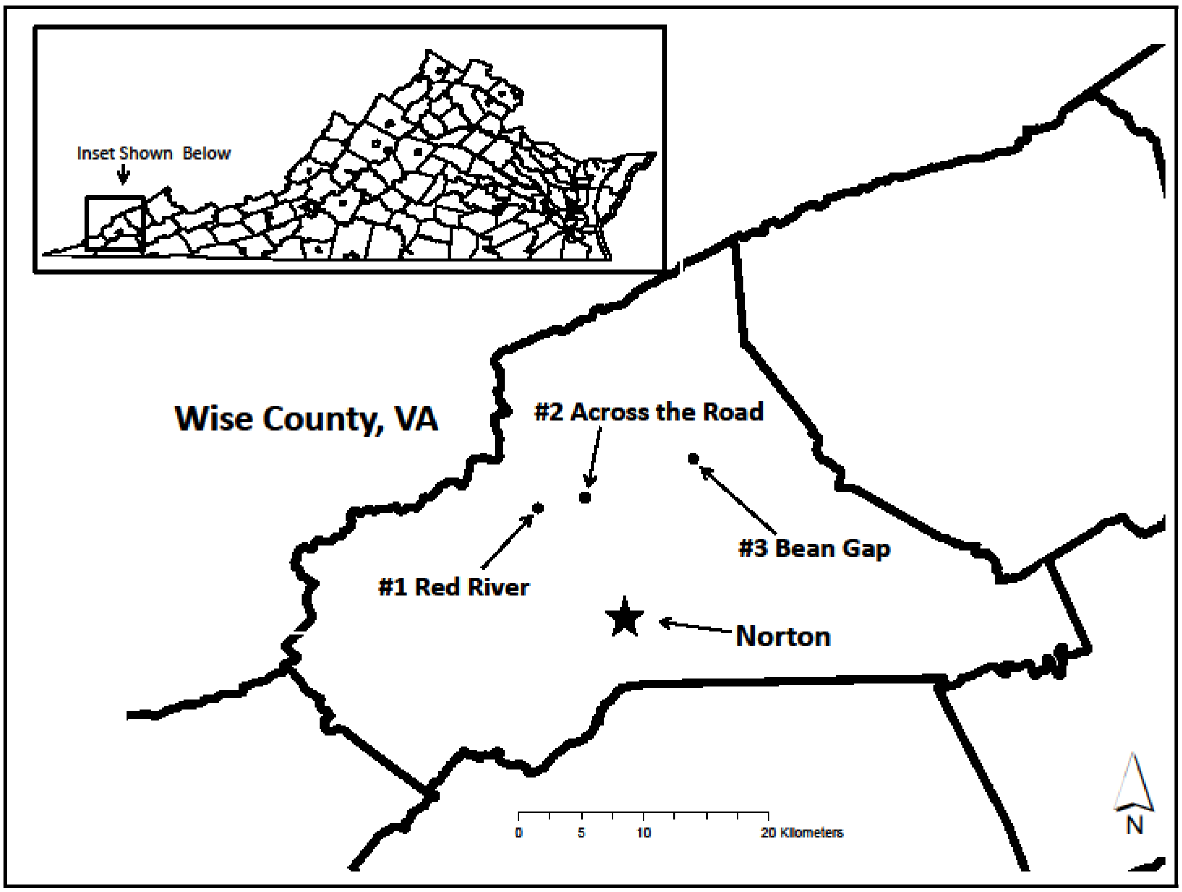

Figure 1. Biomass study block locations on ripped mine sites in Wise County, Virginia.

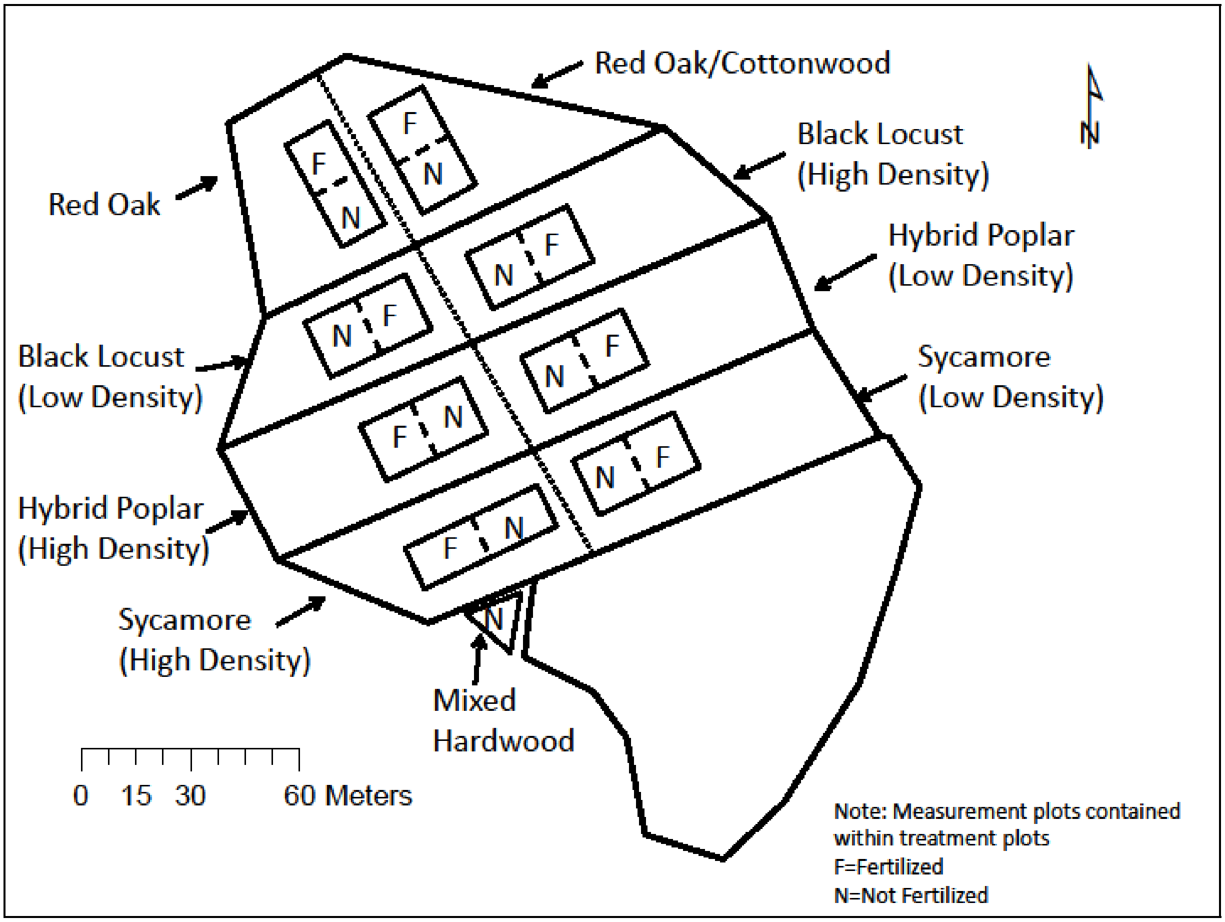

Figure 2 . Example of treatment and measurement plot layout at each of three biomass study sites on ripped mined-sites in Wise County, VA. Included diagram is for Block #3 (Bean Gap).

Harvesting methods drove the rationale for the high and low planting densities. The high-density planting is suitable for harvesting at a young age (5-10 yrs) using a "mowing and chipping" type of harvesting equipment with an operating mechanism that resembles agricultural harvest equipment. The low-density planting would be suitable for harvesting with traditional whole-tree forestry equipment after a longer rotation (16-60 yrs). Within each treatment area and planting density we installed permanent measurement plots of approximately 700 m^2 (Figure 2). A few treatment areas were in non-homogenous land areas, with seasonal or unexpected anthropogenic disturbances, so we reduced the size of these measurement plots to ensure relatively homogenous ground.

In the late spring of both 2008 and 2009, a release spray of 2% glyphosate was used to reduce competition from weeds in a 2 m diameter circle around each of the trees in the treatment area. This spray was hand applied using backpack sprayers. Because of a droughty summer in 2008 and low viability of seedling stock, red oak and cottonwood survival was poor. Therefore, we replanted the red oaks and cottonwoods in the late winter of 2008 to bring the density back to desired levels. In February of 2009 and December of 2009 (after 2008 and 2009 growing seasons), we measured each of the treatment areas for survival, height (height to highest live bud) and ground line diameter (basal diameter). In order to test for fertility constraints on tree growth, we also applied fertilizer to half of each treatment plot, establishing a split plot treatment of fertilization versus no fertilization on all treatments except for the mixed hardwood plots (Figure 2). In December of 2009, 118 milliliters of granular 19:19:19 was applied to the soil surface in a 0.5 m diameter circle around each tree in one half of each measurement plot. Because the nutrients in these fertilizer applications will not be available to the trees until the third growing season, analysis of this treatment is not included in this paper.

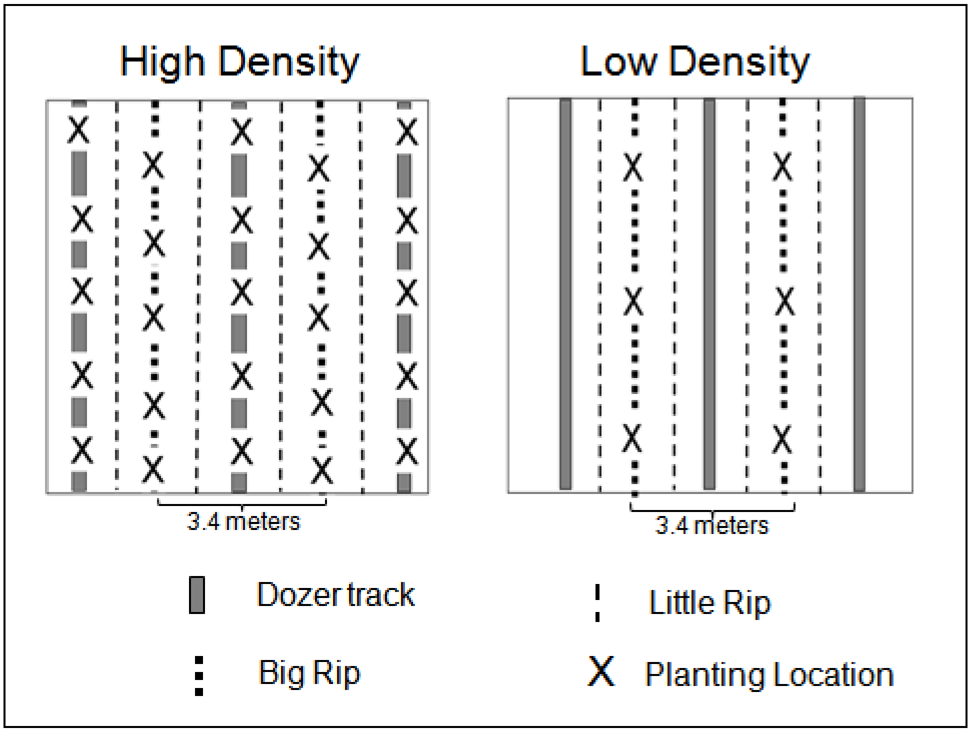

Figure 3. Planting layout for high (1.7 m by 1.7 m) and low (3.4 m by 3.4 m) density planting at biomass study on ripped mined sites in Wise County, VA.

We used a volume index for an estimate of growth that incorporates both height (h) and diameter (d) growth (d^2h). Ground line diameter was used for diameter growth in the calculations. Oven dry wood density was estimated for each species using the Global Wood Density Database (Zanne et al. 2009). Biomass index per tree and per unit area were calculated using these values to give an estimate of dry woody biomass that has been produced in each treatment. Analysis of Variance was used to test for differences in volume growth using the Tukey HSD test for multiple comparisons between our nine planting treatments. Statistical analysis was not conducted on height and diameter measures, due to the young age of the trees. Additional analysis was conducted to test for density, species and block effects on volume growth of sycamore, hybrid poplar and black locust. A 3x3x2 model was used to test for these effects. All analysis was conducted in SAS 9.1. An $\alpha = 0.1$ was used for significance in all analyses.

Table 1. Year one and two stocking, tree diameter and height for tree species and density trial on ripped mine sites in Wise County, Virginia.

Block 1 (Red River)	*n* 2008	*n* 2009	Area (m^2)	Trees ha^{-1} 2009	Mean Basal Diameter (cm)	Mean Height (cm)
High Density Black Locust	210	218	697	3129	2.8	189
Low Density Black Locust	69	72	697	1033	3.2	203
High Density Hybrid Poplar	184	201	697	2885	1.2	98
Low Density Hybrid Poplar	83	93	697	1335	1.5	124
Red Oak	102	113	697	1622	0.7	52
Red Oak/Cottonwood	168	144	697	2067	0.7	49
High Density Sycamore	154	170	697	2440	1.5	88
Low Density Sycamore	40	39	697	560	1.4	80
Mixed Hardwood	46	54	697	775	1.0	65
Block 2 (Across the Road)						
High Density Black Locust	164	163	557	2924	2.3	146
Low Density Black Locust	55	53	465	1141	2.1	134
High Density Hybrid Poplar	130	128	697	1837	1.3	99
Low Density Hybrid Poplar	47	47	697	675	1.1	74
Red Oak	87	84	557	1507	0.6	41
Red Oak/Cottonwood	227	184	697	2641	0.6	46
High Density Sycamore	244	237	697	3401	1.2	61
Low Density Sycamore	62	60	697	861	1.1	51
Mixed Hardwood	20	20	261	766	0.7	37
Block 3 (Bean Gap)						
High Density Black Locust	209	214	697	3071	2.3	158
Low Density Black Locust	84	82	697	1177	3.2	204
High Density Hybrid Poplar	216	212	697	3043	1.3	99
Low Density Hybrid Poplar	87	87	697	1249	1.2	92
Red Oak	101	89	465	1916	0.6	43
Red Oak/Cottonwood	225	172	697	2469	0.7	49
High Density Sycamore	136	150	581	2583	1.4	88
Low Density Sycamore	80	77	697	1105	1.8	107
Mixed Hardwood	29	30	116	2583	1.5	91

Results

Stocking

Our measurements of stocking at year 1 and 2 indicate that planting was successful and survival was adequate (Table 1). However, stocking levels are generally lower than our projected levels of 860 and 3400 trees ha^{-1} for low and high density treatments, respectively, and there is a wide range of stocking at each block and for each treatment. High density second year stocking for hybrid poplar, black locust and sycamore ranged from 1837 to 3401 trees ha^{-1}. While low density stocking for these species ranged from 560 to 1335 trees ha^{-1}. The red oak and red oak/cottonwood treatments were variable between blocks after the second planting (Table 1), with 1507 to 1916 trees ha^{-1} for the red oak and 2067 to 2641 trees ha^{-1} for the red oak/cottonwood treatment. All treatments had generally stable stocking between year 1 and 2, except for the red oak and red oak/cottonwood treatments. These treatments saw declines instocking between year 1 and 2, indicating that the second planting of red oak and cottonwood is also having survival problems. The hardwood treatment stocking was variable between blocks with block 3 having three times higher stocking than blocks 1 and 2. Black locust is the only species that has attained enough mean height growth at all sites and planting densities to be generally above deer browse height and to be generally 'free to grow' (Table 1).

Growth

Black locust had the highest mean per tree volume (index) in both the high and low density treatments (Table 2). Both high and low density black locust had significantly greater mean volume (1311 and 1919 cm^3 respectively) than all other species and density combinations, often by greater than an order of magnitude. The high and low density black locust volumes were not significantly different from each other, but were greater than all other plantings. Hybrid poplar and sycamore had nominally greater volumes than the red oak and red oak/cottonwood, but these differences were not significant. Volume in the low density treatments was nominally greater than in the high density treatments but this difference was not significant.

On a per-tree basis, high and low density black locust had the greatest biomass (786 and 1150 g respectively) (Table 2). Hybrid poplar and sycamore had less biomass per-tree than black locust by an order of magnitude, with red oak and red oak/cottonwood lower by an additional order of magnitude (Table 2). Mixed hardwood was generally intermediate for both mean volume index and biomass per tree. On a per-unit area basis, both high and low density black locust were again greater than all other treatments by an order of magnitude (2.39 and 1.28 Mg ha^{-1} respectively). The high density treatments had approximately double the estimated biomass for black locust, sycamore and hybrid poplar. The red oak and red oak/cottonwood treatments had the lowest biomass on an area basis, by one to two orders of magnitude.

Table 2. Year two volume and biomass index estimates for tree species and density trial on three ripped mine sites in Wise County, Virginia.

Treatment	Mean Volume Index per-tree (cm^3) (n=3)	Mean Stocking (trees ha^{-1})	Oven Dry Wood Density (g cm^3)	Mean Biomass Index per-tree (g)	Mean Biomass Index (Mg ha^{-1})
Black Locust (High Density)	1311a (se=263)	3041	0.60	786	2.39
Black Locust (Low Density)	1917a (se=559)	1117	0.60	1150	1.28
Hybrid Poplar (High Density)	199b (se=12)	2588	0.34	68	0.18
Hybrid Poplar (Low Density)	234b (se=74)	1086	0.34	80	0.09
Sycamore (High Density)	186b (se=74)	2808	0.46	86	0.24
Sycamore (Low Density)	244b (se=100)	842	0.46	112	0.09
Red Oak	24b (se=4)	1682	0.56	13	0.02
Red Oak/Cottonwood	29b (se=4)	2392	0.47	14	0.03
Mixed Hardwood	154b (se=69)	1375	0.47	72	0.10

Note. Volume index with same letters are not significantly different at an $\alpha = 0.1$. Oven dry wood density from Global Wood Density Database (Zanne et al. 2009). Oven dry wood density for mixed hardwood estimated from mean of four hardwood species that are major components of this planting mix.

The results from the multiple comparison procedure showing that mean per-tree volumes for the high and low density treatments were not significantly different for any treatment species is supported by the mixed model ANOVA. Results from the mixed model analysis addressing species, density, block, and species x density effects on mean per tree volume show that only species is significant at $\alpha = 0.1$ (Table 3). Block, density and species x density did not have significant effects on volume index.

Table 3. Analysis of variance of volume index for species and density trial at three ripped mine sites in Wise County, VA.

Source	Degrees of Freedom	Sum of Square	Mean Square Error	F statistic	Pr>F
Block	2	675317	337658	2.05	0.1797
Species	2	9017009	4508504	27.34	<.0001
Density	1	209239	209239	1.27	0.2863
Species x Density	2	343932	171966	1.04	0.3878
Model	7	10245499	1463642	8.88	0.0013
Error	10	1648805	164880		
Total	17	11894304			

Discussion

At year two, we have achieved adequate stocking for all of our species treatments at three sites. These treatments are at a reasonably large scale and allow for a operational scale study of biomass production on mined lands. We have demonstrated the feasibility of establishing woody biomass plantations on previously mined lands with degraded physical properties using ripping as pre-planting treatment. Though there is variability of stocking between the three sites and stocking is lower than expected, all three sites have densities that will allow for intensive biomass production. Additionally, stocking levels are generally consistent between high and low density planting. This will allow us to test if high density plots reach crown closure and produce more biomass earlier than the low density plots. Mean volume growth per tree has yet to be affected by planting density, but we expect to see changes in volume growth per tree as the trees attain crown/root closure and reach the site resource competition stage of forest development. This study will allow us to test for differences in biomass production and above and below ground C sequestration under our two planting densities. It will also allow us to determine how these two planting densities effect biomass production over time to optimize harvest scheduling.

Total volume per tree was clearly greatest for black locust at year two. Combined with good stocking and the highest wood density of any tree in this study, black locust had the greatest biomass production per unit area. Hence, black locust appears to have great potential for woody biomass production on mined- lands that are not fertilized. However, there are concerns that the locust borer *Megacyllene robiniae* (Forst.), a common insect pest of black locust, will have negative effects on tree growth and production over the harvest cycle (Boring and Swank 1984). Our study will allow for examination of this possible effect over the length of the rotation for black locust. An additional concern for end utilization of black locust is its tendency to produce multiple stems. This may increase harvesting costs if the black locust stems are not large enough to use whole tree harvesting equipment, creating concern that standard chopper-type harvesting equipment may not be able to navigate surfaces that have been ripped.

Hybrid poplar and sycamore are intermediate in biomass index at year two. They also have good stocking, which gives them good potential for future volume. However, sycamore has a greater wood density than hybrid poplar (Zanne et al. 2009), which may lead to greater biomass in the future. Sycamore also has the potential to produce sawlogs that may have higher value per volume than pulp or chip materials. Hybrid poplar yields may have been reduced by lack of fertilization at planting, a common cultural treatment in hybrid poplar plantations on un-mined soils. Future growth measurements will reveal if hybrid poplar responds to the recent fertilizer application. The red oak, red oak/cottonwood, and mixed hardwood treatments do not appear to have good potential for biomass production. We successfully established good stocking for these treatments, but they had generally low to moderate volume growth. In addition, the red oak and red/oak cottonwood treatments were also hard to establish and had poor early survival. Even with a second planting, these treatments have moderate stocking and are continuing to lose trees. Our ability to determine if cottonwood can be used to train red oaks may be compromised by poor growth and survival of both species. At year two, the black locust, hybrid poplar, and sycamore treatments appear to be the best suited species for woody biomass production.

Our treatments also showed stocking variability from block to block that may be due to changes in topography, ripping requirements, soil properties, or herbaceous competition effects. This high variability in stocking appears to be common particularly when drastic differences in soil/spoil characteristics between sites is noted (Casselman et al. 2006; Fields Johnson et al. 2008). Growth responses under variable site conditions in mined settings are poorly understood, but must be considered when growing trees for biomass. Future research should be focused on identifying species that do well in various mine soil conditions such as seasonally ponded areas or well drained ridge and shoulder positions. Protocols must also be developed to better schedule final grading of sites with herbaceous planting and tree planting. Biomass production for energy production or C sequestration on mined lands will require advanced understanding of species selection based on site properties and timing of silvicultural activities. This study is an initial step in the understanding of these protocols.

Our estimated rotation ages are 5-10 years for the hybrid poplar, black locust and sycamore biomass crop, with a 16-60 year rotation for the red oak and sycamore saw log crop. Actual rotation ages will be determined by tracking mean annual increment (MAI) over time and harvesting at the peak MAI or the biological rotation age. The hybrid poplar and black locust will be grown strictly for biomass products. Although sycamore is among the fastest growing native Appalachian hardwood species when planted on suitable soils, its growth rate is not as fast as black locust and hybrid poplar. However, its value at rotation age will be much higher if its butt log is used for sawtimber while chipping the rest of the tree for biomass products. At approximately year 12, the interplanted eastern cottonwood will be row-thinned and the red oaks will be left free to grow for a sawtimber rotation. The interplanted treatment will evaluate the silvicultural response of the oaks' stem form due to the presence of the interplanted eastern cottonwoods, as well as total biomass production. This treatment will also help determine the effect of early revenue from biomass on a sawtimber forest enterprise.

This study confronted many of the challenges that need to be addressed to produce biomass on previously mined lands. Though we did not plant herbaceous vegetation, a vigorous cover of volunteer species re-established after ripping. This required the spraying of glyphosate to reduce weed competition around the seedlings, an approach that others have used with success to give planted trees on mined lands higher growth and survival rates (Skousen et al. 2009; Casselman et al. 2006; Ashby 1997; Anderson et al. 1989). Other researchers have had problems with animal browsing (Ashby 1997; Fields-Johnson et al. 2008). After two years, our study has seen little animal damage. At this time, black locust is generally above deer browsing height. All other species, but particularly the red oaks, cottonwoods and mixed hardwoods are still at a height where deer or rodent damage may occur and could set these species back further. Animal browsing control is a factor for successful establishment of biomass plantations and will have to be addressed in many settings. We also included a fertilizer treatment at year 2 to improve soil fertility. The costs of these techniques will vary according to the scale of an operation and the particular conditions at a site. However, successful establishment of woody biomass plantations on past mined lands will often require these additional management techniques and will incur additional costs over the standard FRA protocol on active mine sites.

We plan to track growth over the rotation ages for each treatment with a focus on aboveground biomass production, above and below ground carbon sequestration, and sawtimber production. Analysis will be conducted addressing optimum harvest scheduling timing, as well as per hectare expenditures and possible revenue streams for all of the management scenarios that this study represents.

Conclusions

There are hurdles to overcome in development of techniques to grow woody biomass on both past mined lands or newly mined lands. This study has shown that establishing biomass plantations on past minded lands will require silvicultural practices beyond what is commonly necessary to establish woody vegetation on active mine sites that are being reclaimed under the Surface Mining Control and Reclamation Act of 1977. The foundation for these silvicutural strategies have been developed by foresters and soil scientists, that our now named the Forestry Reclamation Approach (FRA) (Burger 2005). This paper demonstrated the feasibility of installing woody biomass plantations on past mined lands using the FRA, a protocol for reclaiming active mine sites, as a guideline for establishing techniques for soil management, planting, and herbaceous competition control on past mined lands. At year two, our study indicates that black locust, hybrid poplar, and American sycamore are good candidates for biomass plantations on past mined lands. However, future inter-tree competition dynamics, weed competition, species by site interaction effects, and pest impacts will influence which species produces the greatest easily harvestable biomass, in any time-frame.

Acknowledgements

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Stem Form and Fertilizer Effects on Black Locust Biomass Production on Mined Lands

D. Evans, C. Zipper, and J. Burger

Biomass production on mined lands may become a economically viable method for producing relatively carbon neutral fuel stock for energy production or for carbon sequestration. Previous research trials at the Powell River Project (Wise, VA) have indicated that black locust (*Robinia pseudoacacia*) is a species of interest for production of biomass on mined lands, because of its high planting success, rapid growth, and relatively dense wood compared to other more commonly grown biomass species such as hybrid poplar (*Populus trichocarpa* L. x *Populus deltoides*).

Black locust's generally poor stem form is a disadvantage when using it for biomass production. In open growing conditions it can grow multiple stems and could be more costly to harvest or transport compared to single stem trees. This study is intended to test if improved stem form can be achieved through nursery selection. Treatments include black locust seedlings that were culturally selected from parents with superior stem form vs. standard seedlings. We also included an additional fertilizer treatment on half of each of these seedling types to test for limitations due to soil quality.

We included three sites, in Wise County, VA, as replicate blocks for this study. In December, 2007, each site was disked and ripped to till under existing vegetation and to alleviate possible compaction, leaving loose soil material for tree planting and root growth. This was accomplished with a heavy forestland disc harrow used to break up the soil, followed by a second pass to deep till and mound the tree planting row. In the fall of 2009, 2 m diameter circular planting areas were sprayed with 2% glyphosate to remove competing vegetation. In March of 2010 we planted the seedlings at 2.44 m spacing with standard black locust trees as border trees between measurement trees. In order to reduce site quality heterogeneity at each site we included two replications of the seedling type and fertilizer treatment at each site (Figure 1). After planting we applied 118 ml of 19:19:19 granular fertilizer in a 0.3 m circle around each seedling in one half of the treatment plots. Treatment combinations and locations were randomly assigned to each treatment plot using a random number generator. Each treatment comprised a 5 x 5 square plot of trees giving 50 measurement trees for each treatment combination at each block.

Acknowledgements

Thanks to Williams Forestry and Associates for providing the black locust seedlings that were used in this study. Thanks to Alpha Natural Resources, for its support provided for biomass research through Powell River Project. Tnanks to Chris Jackson for his help installing the field trial.

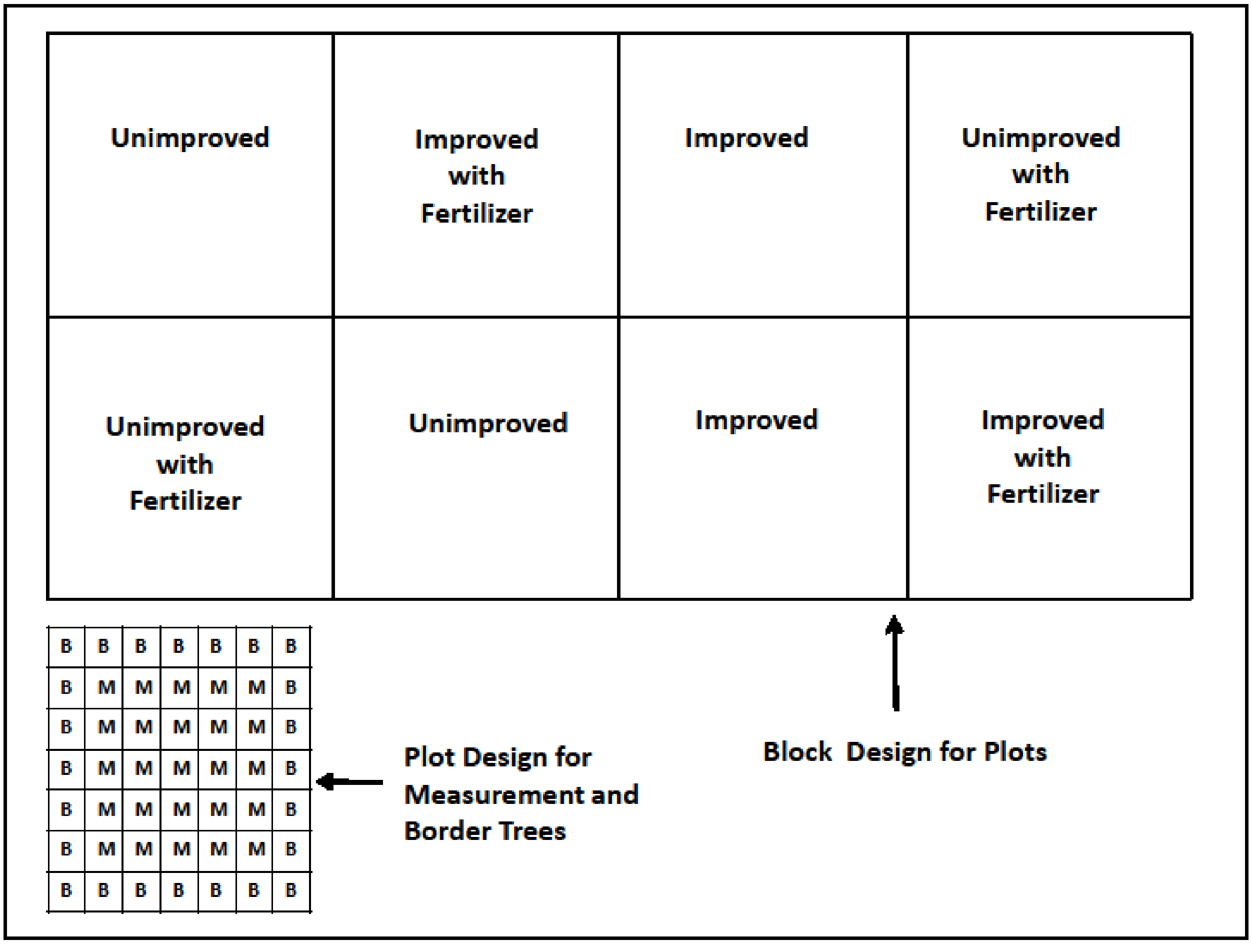

Figure 1. Example of block and plot design for black locust biomass production trial on mined lands in Wise county, VA. Three replications of this study design have been installed; treatments are randomized within each replication.

Hybrid Poplar Biomass Production on Appalachian Reclaimed Mine Land: Year 1 Results of Clone Comparison Trials

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Introduction

Use of Appalachian mined lands for biobased products could help increase energy security, enhance the rural economy, and enhance the environment without displacing land for food production. However, achieving a sustainable biomass-based industry is contingent on the breeding, testing and selection of dedicated perennial cellulosic energy crops specifically for this region.

Poplars (genus Populus) and their hybrids are widely considered to be the premier woody perennial candidate for bioenergy feedstock production on non-mined land sites. We initiated a clonal comparison trial at the Powell River Project Research and Education Center site in 2009. That trial's background and installation were described with greater detail by Brunner et al. (2009). Here, we report first-year results.

Experimental Approach

Our specific objective was to field test 98 clonally replicated genotypes of three major inter-specific taxa. Trials were established in Spring 2009. Ten inch dormant stem cuttings were planted at a 10 feet x 2 feet spacing in a randomized block design with four replications (Figure 1). Clones (Table 1) were blocked by taxa with border rows for each block. Following second year varietal evaluation, above-ground biomass will be harvested and analyzed (2010), and coppicing ability of the varietals assessed (2011). Initial crop production (growth and yield) models will be developed. Results from these trials will be utilized to select varietals for yield verification studies that will confirm selections of superior first-generation commercial varietals.

Table 1. Summary of hybrid poplar clones being evaluated

Taxon		Number of Experimental Clones
Populus x generosa (P. deltoides x P. trichocarpa)	(DT)	33
Populus x Canadensis (P. deltoides x P. nigra)	(DN)	32
Populus deltoides x Populus maximowiczii	(DM)	33
Total		98

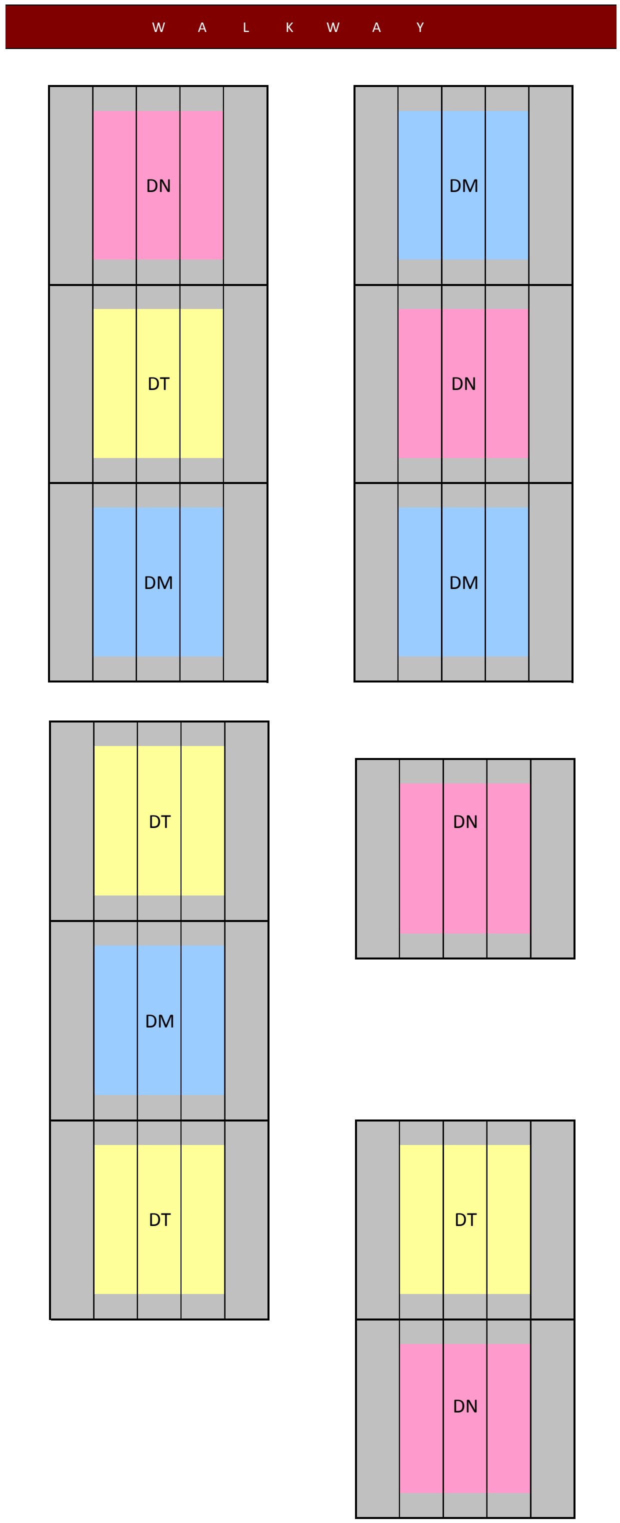

Figure 1. Layout of the hybrid poplar field trial. Gray areas are border plantings (1 row vertical, and two rows at the end of each of the 12 experimental plots). Colored areas are the experimental trials, blocked by taxon as designated in Table 1. Test clones are marked with numbered metal tags.

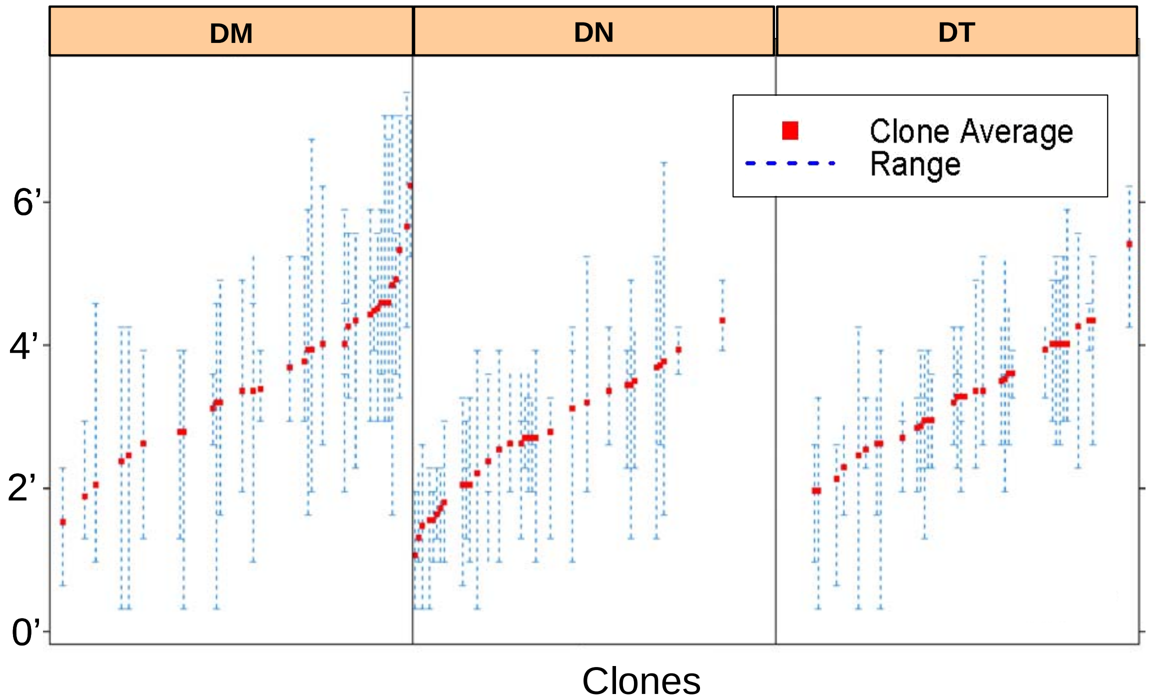

Figure 2. First-year height growth for the hybrid-poplar clones, by taxon.

Preliminary Results

First-year survival was excellent, at >90%. Height growth of surviving trees was highly variable, ranging from <1 foot to >7 feet, with clonal averages ranging from <2 feet to ~6 feet. Average height growth of all clones was >3 feet. Height growth also varied by clone (Figure 2), with *Populus deltoides x Populus maximowiczii* taxon containing the largest number of clones with average height growth > 4 feet. Visual evaluation revealed a minor incidence of poplar leaf rust by the fungus *Melampsora*, as >80% of planted clones were visually rated as having “no” or “light” rust, and <20% were rated for rust as “moderate” or “severe” (Figure 3). Second-year growth appears as excellent (Figure 4) and will be measured at the conclusion of the 2010 growing season.

Acknowledgements

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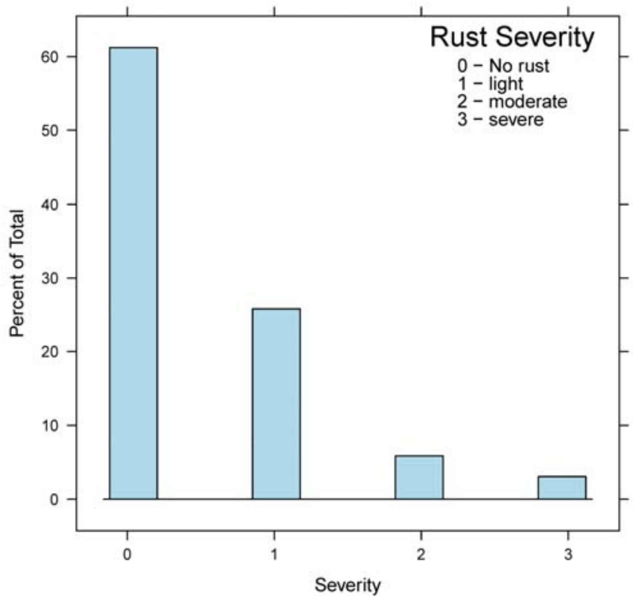

Figure 3. Histogram of poplar leaf rust severity, as evaluated visually at the conclusion of Year 1 growing season.

Figure 4. A section of the hybrid poplar clonal trial during its second year, on 10 August 2010.

RESPONSE OF IMPROVED AMERICAN CHESTNUTS TO PLANTING PRACTICES ON RECLAIMED SURFACE MINED LAND[11]

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Abstract. Hybrids of chestnut that are botanically indistinguishable from American chestnut (*Castanea dentata*) and have the blight-resistance of Chinese chestnut (*Castanea mollissima*) are being developed by the American Chestnut Foundation. Reclamation of mined land in the Appalachians can aid the introduction of these hybrids because of the coincidence of the Appalachian coalfield with the central range of the American chestnut and because of the large areas of land opened up by mining that are available for afforestation. There are questions about whether mined lands can be a suitable habitat for the American chestnut, how the survival of various backcross generations from the breeding program will compare in the field, and the best planting practices to aid establishment. Two experiments were begun to test the performance of several breeding generations of chestnut on reclaimed surface mined land, chestnut compatibility with three ground cover types, and the effect of establishment method. Six breeding generations of chestnuts were direct seeded in 2008 within three different groundcover seeding mixtures: a conventional mine-reclamation mix of tree-competitive legumes and grasses, a tree-compatible mix of less-competitive legumes and grasses, and annual ryegrass only. The 2008 experiment was replicated on three different sites. These trees were planted as nuts in a mix of potting soil, native forest soil and mine soil, and within a tree tube shelter. After two years of growth, the annual ryegrass treatment allowed greater survival (71%) than the conventional tree-competitive seeding mix (50%). In 2009, five breeding generations were planted on four sites, with half planted as unprotected, bare-root seedlings and the other half direct seeded with shelters. After one season, survival of the bare-root seedlings (83%) was higher than that of the direct seeded trees (76%) and the first-year total height of the bare-root seedlings (470 mm) was also greater than that of the planted nuts (347 mm). Survival and growth varied among the various hybrid breeding generations, but none demonstrated consistently superior performance. Labor, time per tree for planting, and supply costs were much greater for the direct-seeded trees than for those planted as bare-root seedlings. Overall, early chestnut survival on a variety of reclaimed mined land is comparable to that of other Appalachian hardwood species. These results suggest that if blight resistance can be effectively conveyed through breeding, reclaimed mined land has potential for use in restoration of the American chestnut as a component of re-established multi-species forests across central Appalachia.

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[1] This paper has been previously published as: C. Fields-Johnson, J. Burger, D. Evans, C. Zipper. 2010. Response of improved American chestnuts to planting practices on reclaimed surface mined land. p. 319-336, in: Proceedings, National Meeting of the American Society of Mining and Reclamation.

Introduction

Background and Rationale

Successful reclamation and afforestation of land surface mined for coal in Appalachia using the Forestry Reclamation Approach (FRA) (Burger et al., 2005) presents the opportunity to also restore American chestnut (*Castanea dentata*), genetically improved to convey blight-resistance, to its native range. Chestnut was an important component of the pre-1950s mixed mesophytic forest (MMF), the restoration of which on mined lands will require chestnut's success in tandem with that of many other native species. The MMF, the Appalachian coal basin, and the core of the American chestnut's former range are all coincident spatially (Figure 1). The MMF type is the oldest and most diverse of the Eastern deciduous forests (Braun 1950) and is being significantly impacted by ongoing surface mining (Wickham et al., 2007; Saylor, 2008).

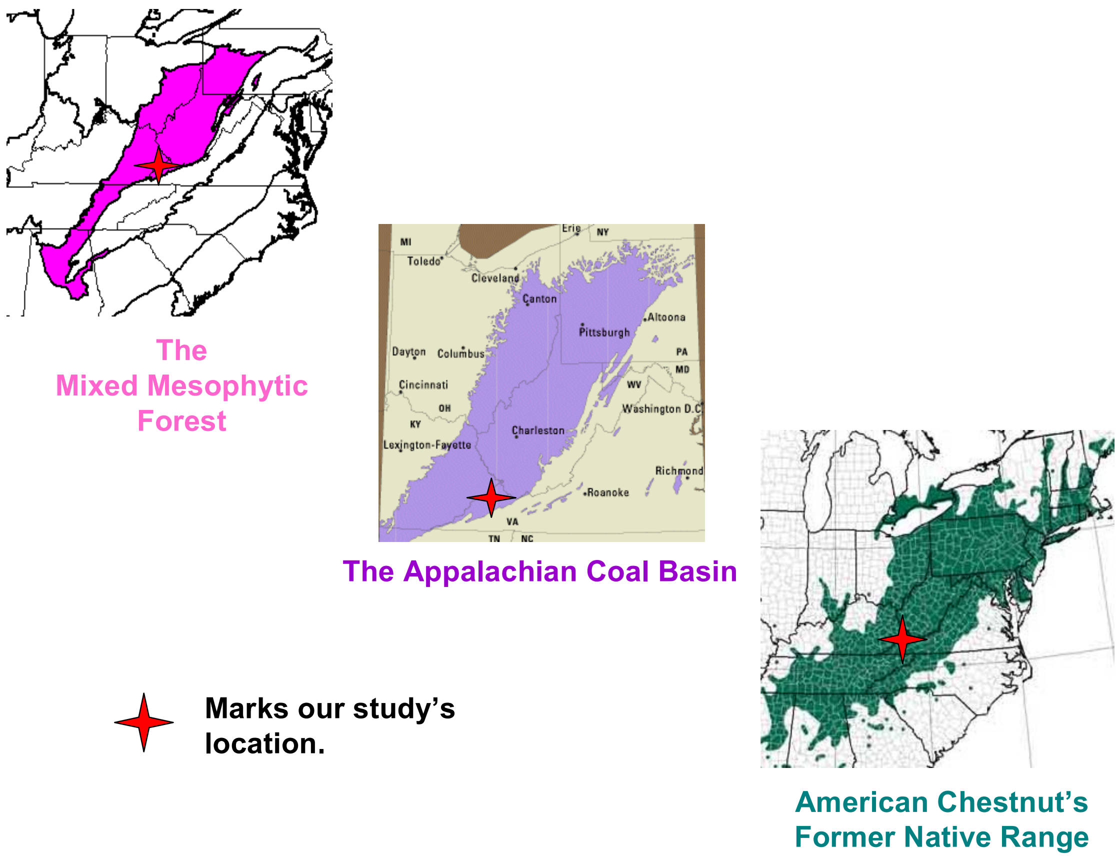
American chestnut was a foundation species of the MMF and also of the Appalachian subsistence culture. The chestnut blight (*Cryphonectria parasitica*) has effectively eliminated chestnut from functioning as it once did in this ecosystem. With efforts to breed a blight-resistant chestnut that is also botanically indistinguishable from American chestnut making progress, it is foreseeable that those efforts' eventual success may provide opportunity to restore the chestnut to the MMF in both form and function. Over 0.6 million hectares of land in the Appalachians has been surface mined for coal (data from United States Office of Surface Mining). These lands, formerly occupied predominantly by MMF, are frequently capped only with barren rock overburden selected for use as a topsoil substitute, without any topsoil, which is used as a starting substrate for revegetation. There is hope that scientific advances and successful implementation of the FRA, which is a five-step process for reforesting mined lands (Burger et al., 2005), within the MMF's range can combine with the successful chestnut breeding programs to achieve the reintroduction of the chestnut on vast expanses of land throughout much of its former range. The mined lands provide an opportunity for chestnut restoration within areas that have little competing forest vegetation which would make full-scale chestnut reintroduction more problematic.

Chestnut Breeding

The American Chestnut Foundation has been breeding American chestnut *(Castanea dentata)* with Chinese chestnut *(Castanea mollisima)* and then using a backcross breeding technique since the early 1980's in order to achieve a blight-resistant hybrid with the form and ecological functions of American chestnut (Hebard, 2001; Diskin et al., 2006; Jacobs, 2007).

The third generation of the third backcross generation (BC3F3) has the botanical characteristics to be classified as American chestnut and is putatively resistant to the blight, but the resemblance of its structural morphology to that of the American chestnut in the long-term has yet to be tested. BC3F3 nuts were first attained in 2005 and will likely be available in larger quantities for reintroduction efforts within one decade (Diskin et al., 2006; Jacobs, 2007).

Figure 1. Study site location, relative to area occupied by mixed mesophytic forest[1], the Appalachian coal basin[2] and the former native range of the American chestnut[3].



1-http://en.wikipedia.org/wiki/File:Appalachia-mixed-mesophytic-forest-map.gif
2-http://www.virgniaplaces.org/graphics/appcoal.gif
3-http://www.patcf.org/images/rangemap2006sff.jpg

American Chestnut as a Foundation Species

Foundation species create and define ecological communities and ecosystems through their structure and function. A small number of strong interactions shape community and ecosystem dynamics in systems dominated by foundation species such as the forests once dominated by American chestnut (Ellison et al., 2005). The loss of foundation species such as the American chestnut has dramatic effects on landscape perceptions as well as on the functioning and stability of ecosystems and all associated biota. The widely ranging environments dominated by American chestnut resulted in alterations to terrestrial and aquatic systems upon its decline. The chestnut is now functionally extinct since its current shrubby form, mostly stump sprouts from trees that succumbed to the blight, produces relatively little leaf area and woody biomass and few nuts. Without this tree fulfilling its historical role, the long term health of the MMF may be compromised (Ellison et al., 2005).

The concept of forest health invokes the concept of ecological integrity as well as the expectation of the presence of all forest relationships and components in a way in which they are fully functional and self-renewing (Oak, 2005). One of the biggest factors in the creation of the present Appalachian forest condition is the chestnut blight. Prior to and during the time of the outbreak of the blight there were heavy forest disturbances from harvesting activity and fire. Following the blight the chestnut trees were eliminated and harvesting-related disturbances began to cease due to changes in policy and management. The net result was the replacement of a chestnut with shade intolerant species, such as oak and hickory. Forest density was also likely changed to a higher-density of canopy trees with dense under-stories (Oak, 2005).

The gypsy moth, to which American chestnut is resistant, favors oaks as a food source and has thus disproportionately defoliated current forest landscapes in comparison to what would have happened pre-blight (Oak, 2005). Therefore the severity of the extensive damage caused by the gypsy moth may be a secondary effect of the chestnut blight. The large decrease in hard mast production with the loss of the chestnut is furthered by oak decline with a subsequent drop in acorn production. This loss of hard mast production has had unknown, but probably significant effects on wildlife and human communities that depended on this food source for sustenance (Oak, 2005). This example highlights how the consequences of the loss of a single foundation species, such as American chestnut, cascade into a series of disruptive consequences for the entire ecosystem. It also demonstrates the need for science-based strategies that lead to the reintroduction of chestnut. We believe that this research will help establish a niche for the American chestnut on lands drastically altered by surface mining that lie within the former range of American chestnut and on lands formerly part of the MMF ecosystem. It can also lead to greater mined land reforestation success.

Silvics, Ecology and Management of American Chestnut

Knowledge of the silvicultural and ecological characteristics of American chestnut is limited because it lost its former ecological role before the advent of modern forest ecology principles (Jacobs, 2007). American chestnut is known to have excellent growth and competitive abilities and can survive in forest understories for prolonged periods before quickly taking advantage of canopy disturbances.. This is characteristic of species that are considered shade tolerant to intermediate in shade tolerance (Jacobs, 2007).

Fast growth and competitiveness of chestnut makes reintroduction in mixed stands with other hardwoods a viable option; however, there is a limited area of sites available for reforestation following logging in the chestnut's range for two reasons: There are policy concerns on public lands regarding the hybrid genetics of improved chestnut, and there are economic concerns on private lands regarding the uncertainty of success at growing chestnut of commercial size (Jacobs, 2007). Afforestation plantings, on surface mines or abandoned agricultural lands may avoid the issues of policy on public land and opportunity costs of private land. Since the American chestnut's original range is all-inclusive of the Appalachian Coal Basin, afforestation of these sites with chestnut following reclamation is a logical means of helping the species recover (Jacobs, 2007).

Most mine soils are derived from rock overburdens that are used as topsoil substitutes. Based on previous research on the influence of chestnut on soils, it appears the species has a disproportionately positive effect on soil quality compared to other native species. In a study of American chestnut trees growing in Wisconsin, outside of the original range of the chestnut and the blight, Rhoades (2007) reported that chestnut stands produced 10-17% higher soil carbon, nitrogen, inorganic nitrogen, net mineralization and nitrification rates than mixed hardwood stands on sandy-loam soils. Total soil carbon, nitrogen and extractable nitrogen were higher under chestnut trees than under mixed hardwoods and soil moisture was somewhat greater. One potential explanation is that the differences caused by chestnut are more apparent on sandy soils whereas they are more buffered and masked on finer-textured soils that have a higher cation exchange capacity, higher amounts of organic matter and higher general nutrient status (Rhoades 2007). American chestnut may have these beneficial effects on sandstone-derived mine soils. Techniques for establishing American chestnut have been explored by past researchers. Phelps et al. (2005) tested the success and effects of different methods of planting American chestnut trees in cleared forest sites. They found that when deer browse activity was absent, tree tube shelters gave trees no advantage in height growth. When there was frequent deer browse activity, tree tube shelters were necessary for establishment. Seedlings were able to successfully compete with other species when the other competing species were cut to ground level mechanically at the time of chestnut planting. Direct seeding was found to be the most cost effective and efficient planting method, but planting of seedlings was found to ensure greater survival, better control over tree placement and enhanced ability to compete with other vegetation. Direct seeded trees did not compete adequately with re-sprouting vegetation that had been cleared (Phelps et al. 2005). On mined land, deer browse and other types of predation would be a concern (Fields-Johnson et al., 2009), but competition from re-sprouting woody plant species commonly found on sites disturbed by logging or fire would be absent on newly reclaimed sites where all vegetation and soil were removed in the process of mining. There are different concerns on mined versus logged land. Experimentation with direct seeding with or without shelters versus planting of seedlings is needed on mined land. Competition from herbaceous species would be a function of the herbaceous species and seeding rate used for mined land erosion control. Experimentation with different seeding prescriptions of herbaceous ground covers with planted chestnut is needed.

Additional Challenges to Chestnut Restoration

Chestnut restoration efforts were begun as early as 1920 by the US Department of Agriculture but failed and were abandoned by the 1960's. The slow process of the dissemination of hypovirus to infected trees has prevented successful treatment of populations of chestnut with hypovirulent strains of the fungus. Other threats to American chestnut that must be overcome include *Phytophthora* root rot (*Phytophthora cinnamomi*), the oriental gall wasp (*Dryocosmus kuriphilus*), and ambrosia beetles (*Xylosandrus crassiulus and Xylosandrus saxeseni*). A limited number of genotypes of American chestnut have provided the basis for the hybrid breeding program and as wild sprouts lose their vigor and die out there is less genetic stock for future breeding. This may undermine restoration due to a lack of adaptation to local environments or to the adaptation of the blight (*Cryphonectria parasitica*) to overcome bred-in resistance genes (Jacobs, 2007).

American chestnut is highly susceptible to *Phytophthora* root rot even when soil compaction and soil moisture are at moderate levels. *Phytophthora* has been isolated from recently reclaimed mine sites with cappings of loose mine spoil (Ward, 2009), indicating that it can be present on mined lands into which chestnut is planted. It will be important to avoid wet and compacted sites that promote *Phytophthora* root rot when planting chestnut trees. Root damage which pre-disposes trees to *Phytophthora,* and transmission of the disease itself to new locations, are both associated with transplanting bare-root seedlings (Rhoades, 2003).

<u>Key Aspects of Reclaimed Mined Land Plantings</u>

Three key aspects of planting chestnuts on mined land are: finding the best hybrid generation of chestnuts to plant, developing the best method of planting the chestnuts themselves, and establishing site conditions through reclamation, including herbaceous groundcover, that are compatible with chestnut establishment. The American Chestnut Foundation (ACF) has been breeding improved hybrid chestnuts from crosses of American chestnut (A) and Chinese chestnut (C). The ACF back-crossed the hybrids with American chestnut three successive times to create three backcrosses with successively higher percentages of American chestnut genes. The first generation of each of the backcrosses was then bred to create an F1, F2 and F3 generation for each backcross (B1, B2 and B3). At the time of this experiment, the F3 generation was available to us for B1 and B2, but the F2 generation was the latest available for B3. Thus three backcross generations (B1F3, B2F3 and B3F2) and two non-hybridized controls for comparison (A and C) were available for our field trials on mined sites. These represent a continuum between American and Chinese chestnut that may produce measurable differences in survival and growth on mined lands.

The pure Americans would be expected to succumb to the chestnut blight and never achieve canopy dominance, though they may repeatedly re-sprout. The pure Chinese would be expected to have a low, spreading growth habit that would also keep them from achieving canopy dominance. Only the hybrids could be expected to have the combination of blight resistance and upright, tall form that would allow them to rise to canopy dominance amidst other native Appalachian hardwoods. This research addresses the initial establishment of these breeding generations. Evaluation of the ultimate success of any of these generations in the mature forest canopy must take place in the long-term

Two potential methods for planting chestnuts are to use young bare-root seedlings or to plant nuts themselves with protective shelters to prevent their consumption by rodents. The bare-root seedling technique requires a year or more of growth in a nursery followed by transplantation. Using nuts for field establishment requires no nursery time, but more intensive activity in the field to increase the probability of successful germination and avoidance of early predation. Current methods for establishing nuts in the field use plastic tree-shelters and steel rebar, both of will remain as non-biodegradable debris if not retrieved, with the rebar acting as a potential hazard with approximately 30 cm protruding vertically from the ground surface if the chestnut tree does not establish successfully. These two methods of establishment were compared for effect on survival and growth, anticipating that results may reveal a preferred establishment method.

It is also well known that herbaceous groundcover influences the survival and growth of trees that are planted on coal surface mines (Burger et al. 2008). Three categories of herbaceous ground cover are 1) those which have been used conventionally in mine reclamation to establish thick, persistent ground cover but also competes with trees; 2) those which are persistent but made up of species which do not compete as vigorously with trees (Burger et al. 2009); and 3) an annual species to create an initial groundcover and then yields to volunteer vegetation (Fields-Johnson et al. 2009, 2010). These three types of ground cover may produce differences in early survival and growth of chestnut; since some ground cover is necessary to prevent early site erosion and to satisfy legal requirements, it is important to know which ground cover type is most compatible with chestnut.

Experimental Objectives

Our goal was to determine which chestnut planting techniques and reclamation strategies can be applied effectively to aid effective American chestnut restoration on reclaimed surface-mined lands. Our objectives were to compare the effects of

1) different breeding generations (Chinese, American, and three generations of backcrosses);
2) three groundcover treatments (conventional, tree-compatible and annual ryegrass only);
3) two planting methods (direct seeding in tree shelters and planting unprotected bare-root seedlings),

on survival and growth of American Chestnut on reclaimed mined land.

Methods and Materials

Two separate experiments were performed to test the effects of planting practices on the survival and growth of American chestnut. The first, begun in 2008, tested the effects of backcross generation selection and ground cover prescription on survival and total height during the 2009 growing season. The second, begun in 2009, tested the effects of backcross generation selection and planting method (direct seeding with tube shelters vs. planting of unprotected bare root seedlings) on survival and total height during the 2009 growing season. These experiments both employed three backcross generations (B1F3, B2F3 and B3F2) as main treatments plus non-hybridized American (A) and Chinese (C) chestnut, from which the hybrids were bred, as non-hybridized controls for comparison.

Both experimental plantings were established on coal-mined areas in the coalfields of southwestern Virginia, USA. Prior to mining, the areas were occupied by mixed mesophytic forest. The area gets approximately 119 cm of precipitation per year and is in plant hardiness zone 6 with average yearly minimum temperatures of -23°C to -18°C. .

2008 American Chestnut Planting: Ground Cover Trial

Six breeding generations of chestnut (2 lines of A and 1 each of C, B1F3, B2F3 and B3F2), provided by the ACF, were planted in mid-March of 2008 with three hydroseed groundcover treatments at three locations (blocks) in southwest Virginia. These sites had all been surface mined for coal and reclaimed in the previous year with steep slopes of approximately 60% and aspects by block of south, east and southeast. The sites were constructed with varying spoil materials to serve as growth media (gray sandstone, brown sandstone, siltstone and some shale). The American chestnuts plantings were established only within the loosely graded treatments of these experimental areas (Fields-Johnson et al., 2010). Each block contained three areas, each roughly 0.4 ha in size and seeded with a different ground cover vegetation: 1) a conventional mix of herbaceous species intended to create >90% ground cover within the first few months of a growing season after seeding, 2) a tree-compatible mix intended to create a moderate level of initial ground cover while eventually covering the soil surfaces fully, and 3) annual ryegrass, intended to create the lowest level of groundcover by planted species while allowing recruitment of native plant species volunteers (Table 1). Within each of the 0.4 ha ground cover treatment areas, approximately 75 nuts were randomly planted among 12 species of Appalachian hardwoods and Eastern white pine (*Pinus strobus*) which were also being established as seedlings on these sites (Fields-Johnson et al., 2010).

Table 1. Hydroseed ground cover treatments for the 2008 chestnut planting.

Annual Ryegrass Only	Rate
Seed Mix:	(kg ha^{-1})
Annual ryegrass (*Lolium multiflorum*)	22
Wood Cellulose Fiber	1680
Tree-Compatible Mix	Rate
Seed Mix:	(kg ha^{-1})
Annual ryegrass (*Lolium multiflorum*)	22
Perennial ryegrass (*Lolium perenne*)	11
Timothy (*Phleum pretense*)	6
Birdsfoot trefoil (*Lotus corniculatus*)	6
Ladino clover (*Trifolium repens*)	3
Weeping Lovegrass (*Eragrostis curvula*)	2
Wood Cellulose Fiber	1680
Conventional Mix	Rate
Seed Mix:	(kg ha^{-1})
Rye grain (*Secale cereale*)	34
Orchardgrass (*Dactylis glomerata*)	22
Perennial ryegrass (*Lolium perenne*)	11
Korean lespedeza (*Lespedeza cuneata*)	6
Birdsfoot trefoil (*Lotus corniculatus)*	6
Ladino clover (*Trifolium repens*)	6
Redtop (*Agrostis gigantea)*	3
Weeping lovegrass (*Eragrostis curvula*)	2
Wood Cellulose Fiber	1680

The conventional ground cover treatment seed mix prescription is one that is commonly applied by a commercial hydroseeding firm on coal mining operations in southwestern Virginia. The tree-compatible mix prescription has been developed through reclamation research using a process of experimentation and observation of many herbaceous species over many years (Burger et al. 2009). Hydroseeding was performed by a commercial contractor using operational procedures, under supervision by the mining firms but using our prescriptions, following final grading of mine spoil. Fertilizer was prescribed for inclusion in all hydroseeding mixtures at an approximate rate of 22 kg ha^{-1} nitrogen (N), 68 kg ha^{-1} phosphorous (P) and 18 kg ha^{-1} potassium. This fertilization prescription for reforestation was developed via experimentation as a way to provide trees ample P without causing excessive herbaceous growth with large amounts of N. Block 1 was hydroseeded in the fall of 2007, Block 2 was hydroseeded in the winter of 2007-2008, and Block 3 was hydroseeded in early spring of 2008. Mining was completed for these sites at different times, hence the staggered hydroseeding schedule.

Chestnut seeds were planted and protected using procedures developed by The American Chestnut Foundation (Figure 2). These procedures involved digging a ~10cm wide x ~20cm deep hole, and filling it with a mix of potting soil, native forest topsoil for biotic inoculation, and on-site mine soil. Seeds were then placed on top of this material and covered with an addition 2-3 cm layer of soil medium. A tree tube (manufactured by Tubex), 6-10 cm in diameter and 38 cm tall, was then placed with its base inserted 2 cm deep into the soil medium and over the seed and moored to a piece of 1-cm thick rebar driven firmly into the ground. Rocks collected on site were piled around the base of each tube to provide additional protection for the buried nut. Nuts were planted in mid-March and germination was assessed in early May. Thereafter, survival, tree height to the highest live bud, and stem diameter at the top of the tree tube were measured in late October – early November at the conclusion of each growing season. Two growing seasons of data were collected for the 2008 planting, with cumulative growth and survival reported here.

Figure 2. Photo of chestnut planting method taken in March of 2008. “Zip ties” inserted through small holes in the tree tube moored the tree tube firmly to the rebar stake.

2009 Chestnut Planting: Planting Method Trial

Five of the six breeding generations of chestnut established in the 2008 experiment, including only one pure American line, were also planted in late March of 2009 on four mined sites in Southwest Virginia with two planting methods. Approximately 180 trees were planted on each site. Two of the planting sites were recently mined areas being actively reclaimed using the FRA. The mine soils were a mix of gray and brown sandstone and siltstone. These two sites were both steep (slopes of approximately 60%) with southerly aspects The third site was a steep area (slope of approximately 60%) with an easterly aspect adjacent to a mine site with surface materials comprised predominantly of soil and weathered sandstone materials which had been re-graded loosely in association with the mining operation. The fourth site was gently sloping, had been mined and reclaimed in the early 1990s with a mix of spoil materials (gray sandstone, brown sandstone and siltstone) and revegetated with grasses, and had been left in an unmanaged condition until December of 2007, when it was treated with a subsoil ripper to relieve soil compaction and then left in an unmanaged state until this planting. On each of the 4 sites, half of the trees were planted as nuts, using methods described for the 2008 chestnut planting; and the other half were planted as one year-old bare root seedlings without any tube shelters or staking. The bare root seedlings were grown in a nursery by the American Chestnut Foundation. Within each block, each row was planted with a single breeding generation; and the direct seeded nuts were alternated with the bare-root seedlings within each row. Survival, tree height to the highest live bud, and stem diameter at ground level, for the bare-root seedlings only, were measured in late October – early November of the first growing season.

Statistical Analysis

Data were analyzed using JMP 7.0 (SAS Institute Inc., Cary NC). Differences in performance characteristics among treatments were determined using a randomized block ANOVA. Tukey-Kramer HSD was used for mean separations ($P < 0.10$). Data from the 2008 and 2009 experiments were analyzed separately. The ground cover trial was designed as a randomized complete block design with ground cover treatment as the main plot and breeding generation as the subplot. The planting method trial was designed as a randomized complete block design with breeding generation as the main plot and planting method as the subplot.

Results

Chestnut survival was significantly greater in the annual ryegrass groundcover than the conventional groundcover, but groundcover type had no significant effect on growth after two growing seasons (Table 2). Planting chestnuts as bare root seedlings as opposed to planting as nuts with tree tubes resulted in significantly greater survival and total height after one growing season (Table 3). There were also significant differences in survival and height among several of the genotypes. Chinese chestnut survival was greater than that of the B2F3 generation, in both sets of plantings, while American and B1F3 survival were also greater than B2F3 in the planting method trial. Chinese chestnuts grew taller than one American chestnut variety for the planting method trial; and they grew taller than all other varieties in the ground cover trial.

Table 2. Cumulative groundcover and genotype effects on survival and total height after two growing seasons for the ground cover trial, with mean separation (Tukey HSD) indicated by different letters beside values within categories.

Groundcover	Survival	α = 0.10	Ht mm	α = 0.10
Annual Ryegrass	71%	a	286	a
Tree Compatible	60%	ab	295	a
Conventional	50%	b	236	a
Genotype				
Chinese	84%	a	373	a
B1F3	73%	ab	352	a
B3F2	65%	abc	276	ab
American 2	58%	abc	203	b
American 1	58%	abc	244	ab
B2F3	48%	bc	273	ab

Table 3. Planting treatment and genotype effects on survival and total height after one growing season for the planting method trial, with mean separation (Tukey HSD) indicated by different letters beside values within categories.

Planting Treatment	Survival	α = 0.10	Ht mm	α = 0.10
Seedlings	83%	a	470	a
Nuts	76%	b	347	b
Genotype				
Chinese	89%	a	740	a
American	87%	a	432	b
B1F3	84%	a	310	c
B3F2	73%	ab	273	c
B2F3	66%	b	287	c

Discussion

The greater survival and first-year height of planted seedlings over planted nuts with tree tubes, combined with the much reduced planting labor and costs, demonstrate that use of bare root seedlings is likely to be a more effective reintroduction technique if tree tube shelters are not needed for protection from herbivory (Phelps et al., 2005). The 38 cm tall shelters used in this experiment were intended to protect nuts and emerging trees from rodents. Taller shelters would be required for protection from deer or livestock.

The seedlings could be planted in less than one minute each with the use of a hoe-dad, whereas direct seeding required over six minutes per seed to dig the hole, add the native soil mix and erect all of the tree protection apparatus, plus additional time to prepare and stage the soils and materials. The cost of labor and supplies for the additional steps of mixing and applying soil and constructing tree shelters when direct seeding caused us to find direct seeding to be more expensive than planting seedlings in contrast to the findings of Phelps et al. (2005). The young trees established as seedlings were taller than those established as nuts, which is not surprising since seedlings had height at the time of planting and are essentially one year older than the trees planted as nuts; the greater height of young trees established as seedlings can be expected to give them an advantage in over-topping herbaceous vegetation during the first growing season, and during subsequent growing seasons if the additional height effect persists as would be expected. Another advantage to the seedling transplant method is that this method is used commonly for re-establishing other native tree species on surface mined lands, providing potential for easier integration of American chestnut within existing mined-land reforestation methods.

Use of only annual ryegrass as a ground cover also improved survival, compared with the conventional ground cover treatment. This was likely due to the lower overall seeding rate and the die-off of the annual rye after 2008 decreasing competition with trees for resources compared to the conventional treatment.

No consistently dominant genotype of hybrid chestnut emerged in these studies, which may change if they differ in sensitivity to Chestnut blight and or in competitive growth form at later growth stages. First-year growth and survival data for the ground cover trial showed few significant differences, between groundcover treatments (Fields-Johnson et al., 2009), compared to second-year results, indicating that treatment effects may continue to diverge with time.

Several other experimental efforts are underway in the Appalachian region testing methods of planting chestnut on reclaimed mined lands, and these results are generally consistent with our findings. French et al. (2008) found that American chestnut direct-seeded on the Cumberland Plateau had greater first-year survival (61.8%) than containerized transplants (51.2%), but height and diameter growth were greater for the containerized transplants. Bare-root seedling transplants survived better than direct-seeded chestnuts in our study, indicating that bare-root seedling transplants may respond differently to out-planting stresses than containerized transplants. Miller et al. (2009) found survival rates of 30%-72% for direct-seeded chestnut in Eastern Tennessee after two months of emergence and growth, and they found that fertilization resulted in a significant decrease in emergence and survival. The trees in our study had generally higher survival rates overall, perhaps due to uniform fertilization applied via hydroseeding rather than to individual trees, but these difference might also have been due to other site or climatic factors. Working in West Virginia, Skousen et al. (2009) found that direct-seeded chestnuts had an overall first-year survival rate of 72%, with 82% for Chinese, 67% for American and 69%-74% for hybrid backcrosses. They found a significant difference in survival between nuts planted with (81%) and without (63%) tree tube shelters, and that the addition of peat to planting holes significantly reduced survival. Our study had a comparable first-year survival rate for nuts planted with tube shelters and comparable patterns of survival by breeding generation.

Our results combined with other studies reported herein can only provide early indications of planting success since all plantings are in early growth stages, but to date they suggest bare-root seedling transplants experience greater survival than direct-seeded chestnuts and direct-seeding results in greater survival than use of containerized transplants. They also suggest direct-seeded chestnuts have greater early survival when protected with tree tube shelters than when planted without shelters, have greater survival when only annual ryegrass is planted as a ground cover than when more competitive and persistent ground covers are used, and have greater survival without additions of peat or direct fertilization of trees at planting than when peat is added or individual trees fertilized.

Survival rates so far in our work (as in some of the other experiments mentioned above) are nominally comparable to those of other mixed native hardwoods planted for research purposes on reclaimed mined land using the Forestry Reclamation Approach. Burger et al. (2008) recorded overall mixed-hardwood survival after 5 years of 69% in research assessing the effects of ground cover control, while Fields-Johnson et al. (2010) recorded survival rates ranging from 71%-75% in 2009 for mixed hardwoods planted as seedlings in association with our ground cover trial of American chestnut, as described above. These early results suggest that, once blight-resistance is effectively conferred, hybrid chestnuts carry the potential for successful introduced throughout American chestnut's former range through reclaimed surface mined land plantings

Conclusions

Planting bare-root chestnut seedlings and hydroseeding annual ryegrass as a sole groundcover were found to be effective ways to improve early chestnut performance on reclaimed surface mined land in the Appalachians. These techniques are also more cost-effective than the alternatives studied. Restoring American chestnut to its native range through plantings on reclaimed mined lands, following the tenets of the Forestry Reclamation Approach, appears promising at this stage so long as blight-resistance is effectively conferred through breeding programs.

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Beef Cattle Cow/Calf Production on Reclaimed Surface Mined Land
Optimizing Production 1997-2010

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Project Summary

The focus of this project is to demonstrate efficient and profitable production of beef cattle on surface mined land in southwestern Virginia. During this cycle an added dimension has been the employment of management-intensive grazing techniques. A herd of forty-two beef cows and ten replacement heifers owned by Penn Virginia Coal are being maintained at the Powell River Project demonstration site in Wise County. The owners have provided pasture; day to day care and management, supplemental feed as needed, and labor to care for the cattle. Virginia Tech, through the co-investigators, has provided advice and assistance with breeding and health management, marketing, maintenance of pasture productivity, record keeping, selection of sires as needed and strategies for obtaining replacements over time. The overriding goal is sustainable beef cattle production with minimum inputs so that costs can be kept low enough to generate profit. .

Introduction

The Powell River Project has successfully demonstrated that reclaimed mine land pastures are well suited to beef cattle production. Data collected between 1980 and 1991 showed that the land and forage resource could be used by beef cows to produce feeder calves at a profit and that this type of use was sustainable with minimal inputs of seed, fertilizer, lime and harvested feeds. Practices defined by Powell River Project programs are now used by producers in the region and feeder calf production is increasing in the region.

A second phase of cattle production, growing and distribution of bred replacement heifers was conducted each year from 1992 through 1995 when forty-five to sixty yearling heifers were grazed at the project site. These heifers were selected from herds outside the region, brought to the site, bred to selected bulls and sold as bred females at auction at the end of the grazing season. Efforts were made to select cattle that would contribute to improving the genetic potential of commercial cattle in the region. Special emphasis was placed on the use of sires selected for calving ease so that the probability of a successful first pregnancy in these virgin heifers could be enhanced.

In the spring of 1996 sixty cow/calf pairs were purchased and placed on the project. The calves were marketed during the fall of 1996. The cows were rebred during the summer of 1996 using a combination of artificial insemination and natural service. In 1997 the decision was made to decrease the herd to approximately 30 cows as additional mining in the area usurped a significant proportion of the grazing lands. These cattle have grazed the existing pastures at the demonstration site along with the purchase of some hay and corn for supplemental winter feeding. Steps have been taken to make full use of the forage resources available on site for year round feeding of the cow herd. Fencing, handling facilities, water supplies and other essential inputs are available on site or have been enhanced as needed.

The operation of this cow/calf program has sufficient scale to generate income and to make efficient use of resources and labor. It is comparable in scope to many similar operations that have been established in the region due in part to prior programs of the PRP. The intent of this report is to demonstrate the most cost effective and profitable management strategies for operation of a beef cow-calf herd on reclaimed surface mined land and to demonstrate ways to enhance the sustainability and profitability of such an enterprise. Techniques for management-intensive grazing have been employed.

Justification and Objectives

Livestock production has been demonstrated to be a productive use of reclaimed land. In recent years, more operators have obtained use of reclaimed land by lease or other means and the number of beef cattle in the coal producing counties has increased as more operators have recognized economic opportunity. The bred heifer project of the PRP aided in this expansion and many of the heifers have gone into herds in Wise, Dickenson and Scott counties. However, it appears that there are opportunities to enhance profitability of these operations by making greater use of the basic forage resource and by employing the best management practices available to beef producers. A primary example of such strategies is the reduced use of harvested feed such as hay by better management of the forage resource to provide near year-round grazing. Also, the quality of the animals can be enhanced by use of improved genetics. Marketing procedures have been improved and greater use of proven management practices and record keeping is beneficial. The employment of most or all of these strategies and procedures has been the objective of this demonstration project. The project has the additional benefit to the coal industry and region by showing that reclaimed land can make an important contribution to the economic life of the community. We are now in the phase of the cattle cycle where numbers are low and prices are quite high. This is a time when significant expansion of beef cattle production might occur in the area as opportunities for profitability are perceived.

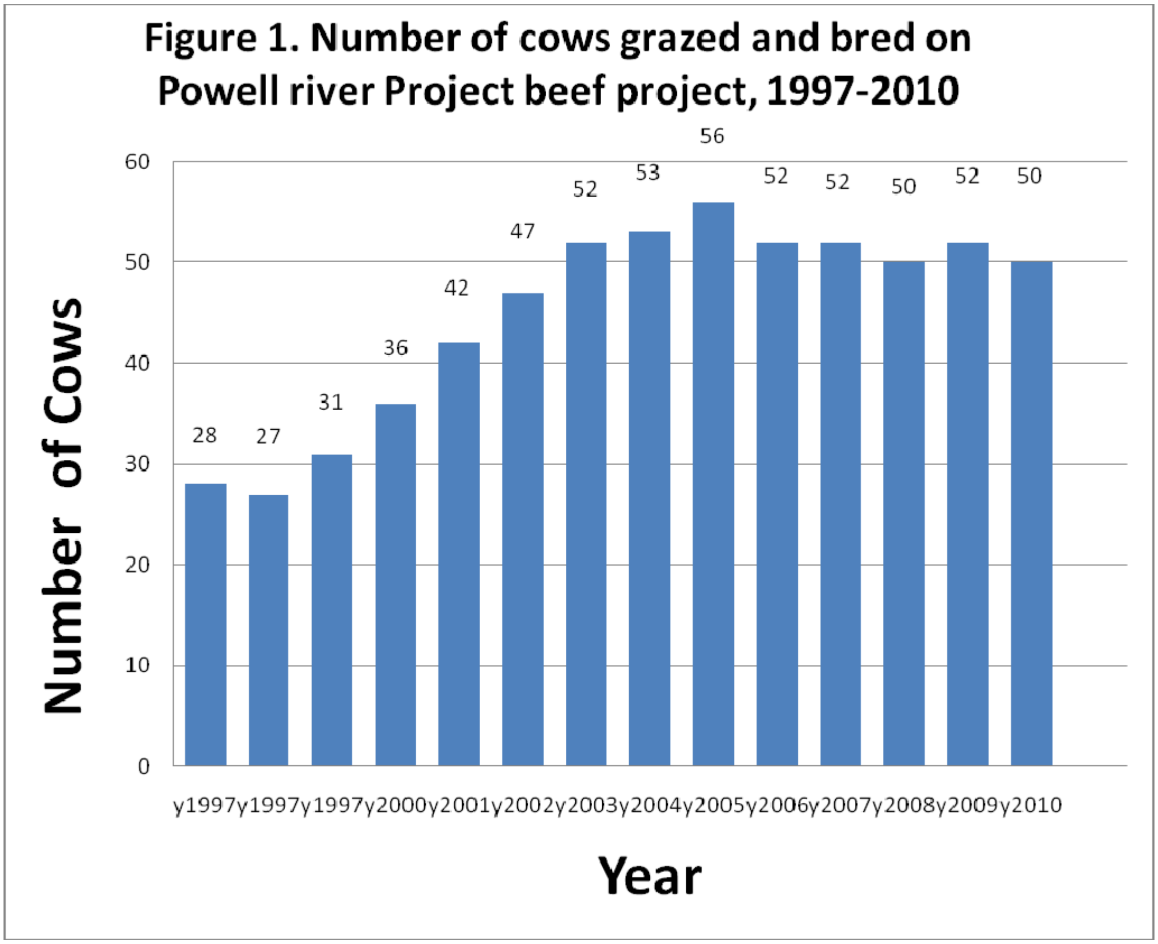
Methods and Procedures

Fifty beef cows and their calves are currently in place at the project site. Seven replacement heifers were weaned and begun development off site over the early winter, then returned to the site to be bred this season. Co-investigators working with Penn Virginia Coal personnel have developed a management and breeding plan for the herd which is being grazed at present. The cows and calves are grazed on the property throughout most of the year with only supplemental feeding when there is severe snow cover during the calving season. Calves are sold between September 15 and November 1. Calving commences about March 1of each year.

Records collected over a fourteen year period of time wee summarized and data is plotted to show trends in cow numbers, success of breeding and calf survival and calf weaning weights.

Results

Figures 1- 3 demonstrate the progress made in production of cattle from 1997 through 2010:



Note that a maximum number of cows was reached in the mid-2000's. It was concluded that running more than about 50 cows wouldresult in the need to buy considerable outside feed for winter needs. It was judged more economical in both dollars and labor to keep the herd numbers near 50 and allow stockpiled grass that grows in the summer and flall to meet most wintering needs.

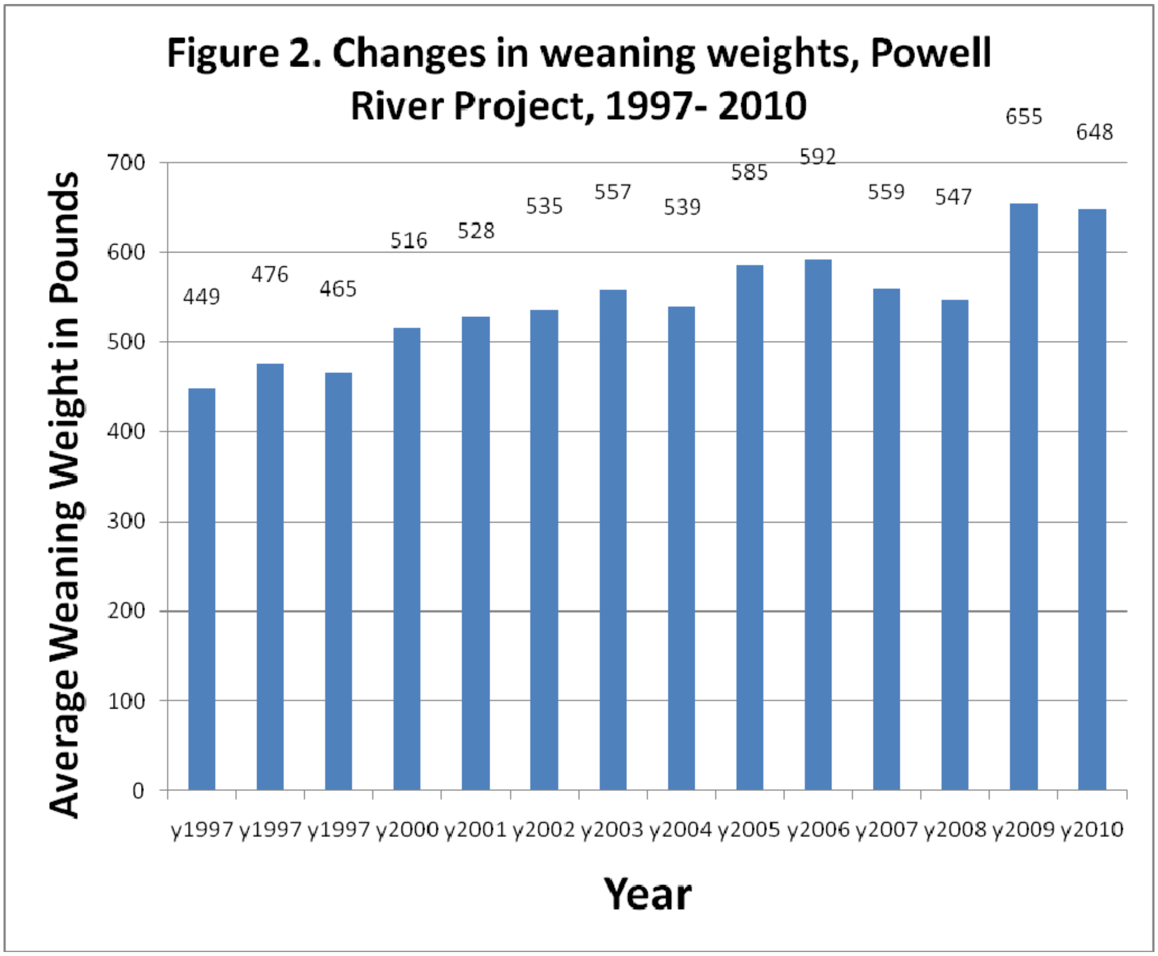
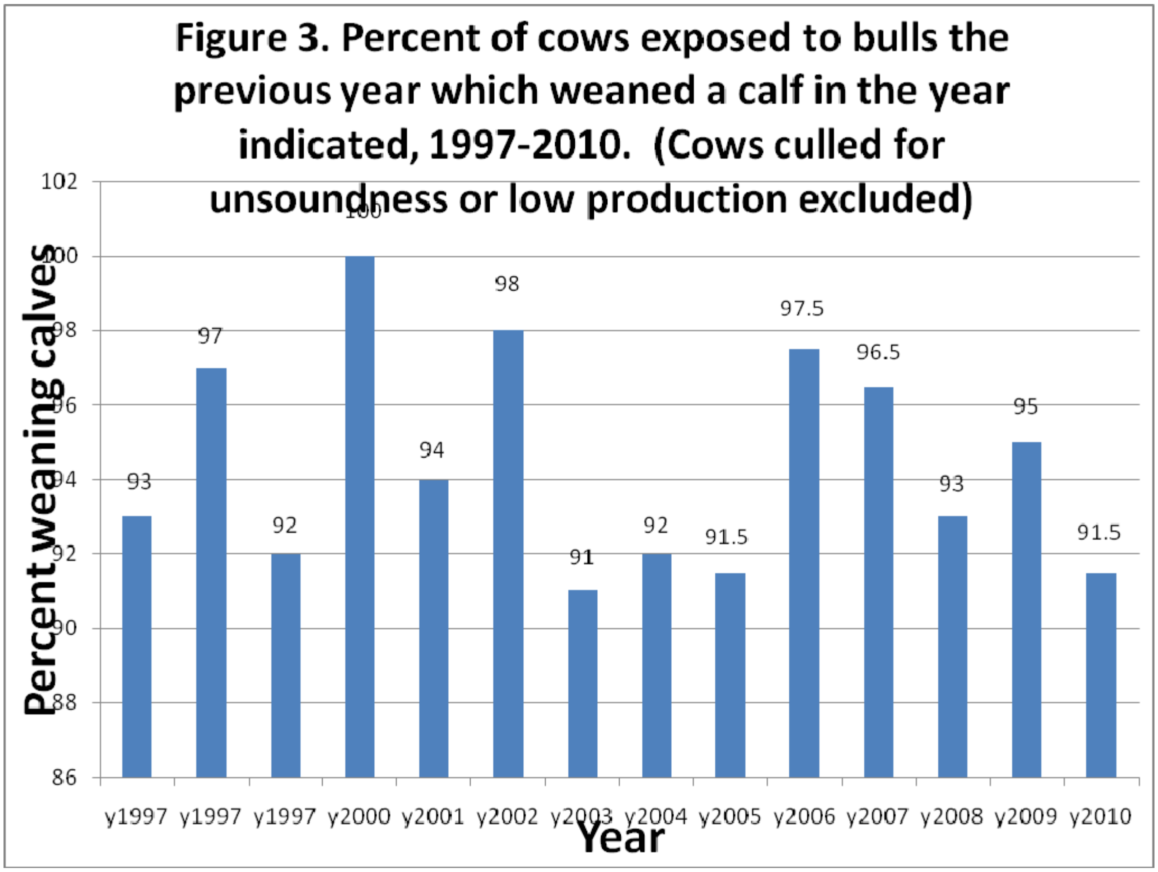


Figure two shoes thaat weaning weights have been continually improving throughour the project. Weights have decreaased during a few years beacause of dry condtions. Use of superior sires, natural and Artificial Insemination, continueal devvelopment of pastures, use of dewormers and growth-promotant implants and better grazing has allowed for the contiula improvement in weaning weights.

Figure three shows that calves weaned per cow exposed has varied somewhat during the project but rates have always been acceptable. When rates have been lover it has typically been because of calf death losses, for example in 2010 because



These figures demonstrate a clear trend in optimizing production of beef on reclaimed strip-mined land. On the same acreage female numbers have been increased, weaning weights have been augmented and although there is some variation, reproductive performance has been maintained.

Major approaches to achieving these ends have included:

- Management intensive grazing principles have been implemented.
- A herd health program to minimize disease losses has been further refined
- By use of artificial insemination and natural service, cows have been bred to sires that have the potential to maximize growth and marketability of the end product - weaned steer and heifer calves.
- A program of fencing, fertilization, overseeding and controlled grazing has been installed to maximize efficient forage production and utilization.

- Water facilities have been improved to provide for high quality fresh water for cows and calves.

Concerns of Toxicity

Concerns have been voiced from various sources concerning the healthiness and healthfulness of cattle raised near a mining site on reclaimed strip-mined pastures. The above summaries of cattle performance certainly refute the idea that these conditions are unfavorable to the production of beef cattle.

The pastures on which the cattle have grazed throughout the thirteen years covered by this report were established by restoring land that had been strip-mined to a relatively level contour. As much top soil as was available was then placed on the surface. Finally, biosolids and wood shavings were placed on the surface to add organic matter to the soil and to increase soil fertility. Finally, a pasture seed mix was applied to the pastures that would grow into plants that were expected to thrive in the environment and provide forage for cows. The seed mix included ladino clover, tall fescue, orchard grass, red clover, sericia lespedeza and autumn olive.

Water sources have been and continue to be a source of challenge for the cattle operation. Originally cavities were scooped into the surface and compacted designed to become pods from water runoff. Many of these eventually filled with silt and became unusable. A tank was installed associated with the barn on the site to capture rain water and this was routed to the waterers for a few years but this proved unsustainable. For several years water was pumped from water filled deep mines below but eventually the electric source was removed. Now, when ponds are dry water is hauled from the river and placed in watering tanks. There is a plan to drill a well to provide long-term, dependable, high quality water for the cattle.

Potential sources of contamination of cattle are the biosolids that were applied to the land when the pastures were established. As the area has continued to be mined, especially from above levels on the site, blasting has occurred and trucks have hauled coal on a road that runs adjacent to all cattle pastures. There is the potential that dust from the coal trucks or from blasting used to loosen rock and coal on adjacent mining sites has contaminated the forages that cattle have consumed. It should be noted that roads are kept aggressively sprinkled to minimize dust. Water contaminated by mining might be an additional source of contamination.

There is a plan to harvest tissue from cows that are culled this fall so that analysis can be made for contaminants such as lead, arsenic, molybdenum, cadmium, etc.

Even though it is obvious that levels of toxins have not been sufficient to hurt cattle production, it must be assured that toxic substances that might be harmful for those who consume these cattle as beef have not accumulated.

Summary

This demonstration is a highly visible example to area producers of what can be accomplished using available information in the most effective manner. It is hoped this will lead other producers to adopt techniques to enhance the productivity of their beef cattle operations.

Cows grazing reclaimed surface-mined land can be very productive. If stocking rates are kept moderate, very little harvested feed is utilized in the production of high quality feeder cattle.

Determining Water Quality Criteria for Total Dissolved Solids in Streams of Southwestern Virginia

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Project Summary

This research is being conducted for the purpose of recommending total dissolved solids (TDS) criteria for developing water quality standards with potential application to Virginia's coalfield streams. Results will be used to develop a statistical model that predicts a Virginia Stream Condition Index (VA SCI) score as a function of TDS concentrations and/or component ion contributions to TDS at locations where non-TDS anthropogenic stressor effects are not evident. The statistical model will be used to recommend a criterion that can be used to develop a TDS water quality standard that would protect aquatic life within the study region. Research methods are as follows:

1. Identify freshwater stream research sites that have elevated (i.e., above reference) TDS concentrations but appear to be otherwise relatively unaffected by non-TDS stressor effects.
2. Identify freshwater stream research sites that can serve as reference locations for assessment of TDS effects on the VA SCI.
3. At each research site, sample benthic macroinvertebrates.
4. Characterize non-TDS stressor and other benthic macroinvertebrate community influences at all research sites by sampling habitat elements and water quality.
5. Analyze data to determine a potential TDS criterion.

Progress Update

Progress since our previous report submitted in April 2010 has been on schedule and within budget. Sample processing has been completed for Fall 2009 samples. Chemical analyses for TDS, alkalinity, major ions, and trace metals have been completed. Benthic macroinvertebrate samples have been sorted and organisms have been identified. Fall 2009 VA SCI scores have been calculated for each site. Chemical and biological data for Fall 2009 are summarized herein.

Biological and chemical samples were collected from 28 sites during the Spring 2010 index period, which included three additional reference sites selected during winter 2010. Chemical and biological analyses are ongoing for Spring 2010 samples. Spring 2010 data will be included in the final report.

Mr. Timpano has been certified in genus-level taxonomy (Eastern EPT) by the North American Benthological Society.

This report presents a summation of Fall 2009 data, activities of the past four months, and plans for the remainder of the project, to be concluded in December 2010. Earlier results were reported by Timpano et al. (2009).

Site Selection

Three additional reference sites were sampled during the Spring 2010 index period. Reference sites represent the least disturbed condition that we could locate in the Virginia coalfield region, while remaining as comparable as possible to test sites with regard to factors such as geology, topology, and hydrology. Initial candidates were selected through review of landuse data, aerial photography, as well as water quality and permit data provided by DMME. Candidate reference sites were then visited and evaluated for inclusion in the study.

Presentations

A manuscript was accepted for, and oral presentation made during, technical sessions of the 2010 National Meeting of the American Society of Mining and Reclamation (ASMR), held in Pittsburgh, Pennsylvania in June 2010. The manuscript was published in the 2010 ASMR conference proceedings. The paper presented preliminary results concerning TDS relationships to specific benthic macroinvertebrate metrics, but did not draw conclusions regarding potential VA SCI impairment thresholds or criteria. The paper is attached as an Appendix to this report.

An abstract was accepted for presentation of a poster at the 2010 meeting of the Society for Environmental Toxicology and Chemistry to be held in Portland, Oregon in November 2010. The poster will present our findings regarding TDS ionic composition as well as an analysis of biological response to TDS component ions. That abstract is also attached.

Future Plans

The plans for the next four months include finalizing analyses of chemical and biological samples from Spring 2010, data analysis, and final report preparation. The final report will contain all raw data collected during this study, as well as complete analysis of those data. The report is scheduled for submission by December 31, 2010.

Reference::

A. Timpano, S. Schoenholtz, D. Soucek, C. Zipper. 2009. Effects of Total Dissolved Solids in Streams of Southwestern Virginia. p. 82-94, in: 2009 Powell River Project Research and Education Program Reports. http://www.cses.vt.edu/PRP/Reports_09/Reports_09.html

Data Summary – Fall 2009

Table A-1. Site Information - Fall 2009

Stream	Station ID	Type	Order	Dominant Geology	County	Lat	Long
Burns Creek (ref)	BUR	Ref	2	Lee	Wise	36.929	-82.535
Clear Creek (ref)	CLE	Ref	2	Mississippian	Wise	36.929	-82.589
Eastland Creek (Ref)	EAS	Ref	1	Mississippian	Wise	36.917	-82.593
Birchfield Creek	BIR	Test	2	Wise	Wise	37.036	-82.575
Callahan Creek West Fork	CAW	Test	1	Wise	Wise	36.980	-82.797
Fawn Branch	FAW	Test	1	Wise	Lee	36.811	-83.080
Fryingpan Creek	FRY	Test	2	Norton	Dickenson	37.060	-82.218
Fryingpan Creek Right Fork	RFF	Test	2	Norton	Dickenson	37.060	-82.220
Gin Creek	GIN	Test	3	Wise	Lee	36.836	-83.055
Grape Branch	GRA	Test	2	Norton	Buchanan	37.257	-82.007
Hurricane Fork	HUR	Test	2	Norton	Buchanan	37.400	-82.067
Jess Fork	JES	Test	2	Wise	Buchanan	37.295	-82.219
Laurel Branch	LAB	Test	2	Norton	Russell	37.014	-82.205
Laurel Fork	LAU	Test	2	Wise	Wise	36.874	-82.825
Mill Branch Left Fork	MIL	Test	2	Wise	Wise	36.927	-82.747
Powell River	POW	Test	1	Wise	Wise	37.013	-82.697
Race Fork UT	RAC	Test	1	Norton	Buchanan	37.427	-82.050
Roll Pone Branch	ROL	Test	1	Norton	Russell	37.014	-82.195
Spring Branch	SPR	Test	1	Norton	Buchanan	37.434	-82.046
Cane Branch	CAN	Test	1	Wise	Dickenson	37.160	-82.547
Kelly Branch	KEL	Test	2	Wise	Wise	36.935	-82.792
Kelly Branch UT	KUT	Test	1	Wise	Wise	36.936	-82.792
Richey Branch	RIC	Test	2	Wise	Wise	37.036	-82.546
Richey Branch UT	RUT	Test	1	Wise	Wise	37.037	-82.544
Spruce Pine Creek	SPC	Test	2	Norton	Buchanan	37.261	-81.922

Table A-2. Field Physicochemical Parameters - Fall 2009

Stream	**Temp (°C)**	**pH (SU)**	**D.O. (mg/L)**	**Cond. (μS/cm)**
Burns Creek (ref)	13.62	6.5	8.36	21
Clear Creek (ref)	13.81	7.25	8.39	20
Eastland Creek (ref)	13.79	7.15	7.69	22
Birchfield Creek	14.38	7.67	8.45	647
Callahan Creek West Fork	7.81	7.31	10.75	292
Cane Branch	10.01	7.96	9.13	1462
Fawn Branch	9.55	7.49	8.52	281
Fryingpan Creek	13.08	7.62	9.05	402
Fryingpan Creek Right Fork	12.9	7.41	8.37	340
Gin Creek	11.14	8.07	9.21	656
Grape Branch	8.35	7.27	10.12	339
Hurricane Fork	7.31	7.21	11.06	383
Jess Fork	6.74	7.22	11.35	682
Kelly Branch	11.67	7.3	9.34	873
Kelly Branch UT	12.71	7.96	9.69	1366
Laurel Branch	13.01	7.63	9.19	784
Laurel Fork	7.04	7.23	10.15	20
Mill Branch Left Fork	15.00	7.37	8.42	845
Powell River	15.08	7.46	8.93	1087
Race Fork UT	9.31	7.25	9.69	450
Richey Branch	13.49	7.93	8.49	1670
Richey Branch UT	12.85	7.63	8.77	545
Roll Pone Branch	12.97	7.2	8.77	652
Spring Branch	7.68	7.3	10.69	274
Spruce Pine Creek	9.01	7.79	10.05	468

Table A-3. Rapid Bioassessment Protocol Habitat Scores - Fall 2009

Stream	**Substrate/Cover**	**Embeddedness**	**Velocity/Depth**	**Sediment Dep.**	**Flow**	**Channel Alt.**	**Riffle Freq.**	**Bank Stability L**	**Bank Stability R**	**Veg. Protection L**	**Veg. Protection R**	**Rip. Veg. Width L**	**Rip. Veg. Width R**	**Total**
Burns Creek (ref)	19	18	17	17	18	20	17	9	10	10	10	10	10	**185**
Clear Creek (ref)	19	17	19	16	20	20	20	9	10	10	10	10	10	**190**
Eastland Creek (ref)	19	17	17	17	17	20	20	9	9	10	10	10	10	**185**
Birchfield Creek	16	13	15	12	18	20	18	7	7	8	8	10	8	**160**
Callahan Creek West Fork	16	13	16	13	18	20	18	9	9	10	10	10	10	**172**
Fawn Branch	17	13	16	13	18	20	18	9	9	9	9	10	9	**170**
Fryingpan Creek	18	15	16	14	17	20	19	9	9	10	10	10	10	**177**
Fryingpan Creek Right Fork	18	16	16	14	15	20	18	8	8	10	10	10	9	**172**
Gin Creek	16	12	16	12	18	20	18	8	8	8	8	10	7	**161**
Grape Branch	17	14	16	13	18	20	16	9	9	9	9	10	10	**170**
Hurricane Fork	16	11	15	11	15	20	18	8	8	9	9	10	10	**160**
Jess Fork	15	12	15	11	18	20	17	7	7	7	7	9	7	**152**
Laurel Branch	16	13	15	12	20	20	16	8	8	8	6	10	6	**158**
Laurel Fork	20	15	17	14	20	20	19	9	9	10	10	10	10	**183**
Mill Branch Left Fork	17	13	13	12	16	20	18	8	7	10	10	10	9	**163**
Powell River	19	15	15	14	20	20	20	7	9	10	10	10	10	**179**
Race Fork UT	18	13	14	12	16	20	18	8	8	9	9	10	10	**165**
Roll Pone Branch	17	12	15	12	16	20	19	8	8	9	9	10	10	**165**
Spring Branch	17	13	16	12	17	20	17	7	8	7	8	10	10	**162**
Cane Branch	17	14	17	13	18	20	17	7	7	8	8	10	10	**166**
Kelly Branch	17	13	18	12	20	20	17	10	10	10	9	10	10	**176**
Kelly Branch UT	18	16	17	14	18	20	19	6	6	9	9	10	10	**172**
Richey Branch	17	14	17	13	18	20	17	7	7	9	9	10	10	**168**
Richey Branch UT	17	13	17	12	18	20	17	6	6	10	10	10	10	**166**
Spruce Pine Creek	17	15	17	13	17	20	18	10	10	10	10	10	10	**177**

Table A-4. Total Dissolved Solids, Anions, and Alkalinity - Fall 2009

Stream	TDS (mg/L)	Anions (mg/L)		Alkalinity (mg/L as $CaCO_3$)		
		Cl^-	SO_4^{2-}	Total	CO_3^{2-}	HCO_3^-
Burns Creek (ref)	12	2.0	4.2	1.2		1.2
Clear Creek (ref)	14	0.5	3.1	6.3		6.3
Eastland Creek (ref)	10	0.4	2.8	8.0		8.0
Birchfield Creek	410	8.7	220.7	120.1		120.1
Callahan Creek West Fork	187	0.7	88.2	68.4		68.4
Cane Branch	1108	5.1	679.4	202.0		202.0
Fawn Branch	164	2.2	67.4	81.5		81.5
Fryingpan Creek	263	11.2	100.9	93.1		93.1
Fryingpan Creek Right Fork	218	5.9	76.7	90.5		90.5
Gin Creek	411	8.5	112.9	232.9	3.4	229.5
Grape Branch	202	4.2	119.7	51.6		51.6
Hurricane Fork	258	1.1	166.5	34.2		34.2
Jess Fork	493	1.1	340.4	49.3		49.3
Kelly Branch	615	1.8	412.7	88.0		88.0
Kelly Branch UT	1021	1.9	629.3	173.1		173.1
Laurel Branch	553	3.9	282.7	124.7		124.7
Laurel Fork	33	0.5	4.3	6.0		6.0
Mill Branch Left Fork	588	1.1	350.8	133.6		133.6
Powell River	751	0.5	477.2	126.7		126.7
Race Fork UT	273	1.1	163.0	81.4		81.4
Richey Branch	1378	5.8	849.0	190.8		190.8
Richey Branch UT	388	4.6	219.4	75.1		75.1
Roll Pone Branch	462	3.2	272.4	83.4		83.4
Spring Branch	156	0.8	92.3	53.3		53.3
Spruce Pine Creek	281	3.4	108.2	142.9		142.9

Table A-5. Major Cations - Fall 2009

Stream	Major Cations (mg/L)			
	K	Na	Ca	Mg
Burns Creek (ref)	0.3	1.7	1.2	0.5
Clear Creek (ref)	0.4	0.6	2.1	0.6
Eastland Creek (ref)	0.4	0.6	2.8	0.7
Birchfield Creek	4.2	19.4	63.2	46.3
Callahan Creek West Fork	2.1	6.8	33.1	14.9
Cane Branch	7.4	76.8	141.8	97.0
Fawn Branch	2.2	10.9	29.9	13.7
Fryingpan Creek	2.4	27.0	37.2	18.0
Fryingpan Creek Right Fork	2.4	29.1	29.0	11.9
Gin Creek	3.7	117.1	28.7	11.8
Grape Branch	2.0	21.0	32.1	13.3
Hurricane Fork	2.5	7.7	36.0	27.3
Jess Fork	3.3	9.9	81.6	45.2
Kelly Branch	4.9	14.1	100.3	59.6
Kelly Branch UT	7.6	55.3	151.8	82.4
Laurel Branch	4.3	46.5	82.0	41.9
Laurel Fork	0.6	0.7	1.5	1.1
Mill Branch Left Fork	5.1	15.2	104.4	49.6
Powell River	4.5	9.7	122.7	72.0
Race Fork UT	2.5	20.7	46.4	23.8
Richey Branch	6.5	14.6	183.9	160.6
Richey Branch UT	4.2	5.5	46.4	50.1
Roll Pone Branch	3.9	16.0	76.5	39.6
Spring Branch	1.9	3.8	27.6	18.3
Spruce Pine Creek	1.9	53.8	37.0	15.7

Table A-6. Trace Metals - Fall 2009

Stream	Trace Metals (μg/L)					
	Al	**Cu**	**Fe**	**Mn**	**Se**	**Zn**
Burns Creek (ref)	11.3	< 12.9	< 64.9	6.7	< 16.1	18.1
Clear Creek (ref)	35.1	< 12.9	< 64.9	12.2	< 16.1	10.9
Eastland Creek (ref)	< 8.6	< 12.9	< 64.9	5.1	< 16.1	10.3
Birchfield Creek	< 8.6	< 12.9	< 64.9	32.9	< 16.1	12.0
Callahan Creek West Fork	9.4	< 12.9	< 64.9	6.3	< 16.1	12.6
Cane Branch	25.8	< 12.9	< 64.9	86.7	< 16.1	10.3
Fawn Branch	< 8.6	< 12.9	< 64.9	8.1	< 16.1	12.8
Fryingpan Creek	< 8.6	< 12.9	< 64.9	6.6	< 16.1	10.0
Fryingpan Creek Right Fork	< 8.6	< 12.9	< 64.9	10.4	< 16.1	10.2
Gin Creek	< 8.6	< 12.9	64.9	7.4	< 16.1	13.7
Grape Branch	< 8.6	< 12.9	< 64.9	6.3	< 16.1	11.4
Hurricane Fork	8.7	< 12.9	< 64.9	17.2	< 16.1	11.6
Jess Fork	20.0	< 12.9	< 64.9	12.7	< 16.1	16.7
Kelly Branch	< 8.6	< 12.9	< 64.9	9.5	< 16.1	10.6
Kelly Branch UT	17.2	< 12.9	< 64.9	12.2	22.9	11.3
Laurel Branch	< 8.6	< 12.9	< 64.9	11.2	14.6	11.1
Laurel Fork	12.4	< 12.9	80.8	10.6	< 16.1	15.9
Mill Branch Left Fork	15.3	< 12.9	71.3	93.0	17.2	13.5
Powell River	11.4	< 12.9	< 64.9	14.1	16.6	17.4
Race Fork UT	< 8.6	< 12.9	< 64.9	6.5	< 16.1	11.3
Richey Branch	36.4	< 12.9	< 64.9	19.7	< 16.1	10.2
Richey Branch UT	9.9	< 12.9	< 64.9	11.7	< 16.1	10.5
Roll Pone Branch	< 8.6	< 12.9	< 64.9	5.8	< 16.1	11.1
Spring Branch	30.7	< 12.9	69.9	8.7	< 16.1	10.8
Spruce Pine Creek	< 8.6	< 12.9	< 64.9	14.7	< 16.1	10.7

Table A-7. Virginia SCI Metrics and Final Score (100 organism sample) – Fall 2009

Stream	# Taxa	# EPT Taxa	% E	% PT-Hyd.	% Scrapers	% Chiron.	% 2 Dom.	HBI	SCI Score
Burns Creek (ref)	15	8	7.5	34.0	4.7	16.0	51.9	4.2	62.1
Clear Creek (ref)	20	16	31.0	19.0	22.4	15.5	40.5	4.2	74.3
Eastland Creek (ref)	15	11	17.4	22.7	10.6	3.8	60.6	4.3	64.8
Birchfield Creek	9	5	0.0	76.0	0.0	1.0	82.7	2.2	51.3
Callahan Creek West Fork	17	12	19.8	42.6	17.8	9.9	38.6	3.2	77.8
Fawn Branch	17	12	30.1	56.6	9.7	6.2	54.0	2.1	75.7
Fryingpan Creek	15	12	0.9	60.4	9.0	9.0	71.2	3.0	65.0
Fryingpan Creek Right Fork	9	5	0.0	83.3	1.9	7.4	86.1	1.9	50.3
Gin Creek	11	7	1.8	56.0	0.9	8.3	83.5	3.2	54.2
Grape Branch	13	9	1.9	41.3	1.0	14.4	67.3	4.2	58.1
Hurricane Fork	23	16	9.0	49.5	6.3	9.9	43.2	3.4	74.5
Jess Fork	11	8	2.6	17.9	0.0	23.9	73.5	5.2	45.2
Laurel Branch	13	7	5.0	37.1	1.4	11.4	68.6	4.0	57.0
Laurel Fork	13	11	18.5	48.1	11.1	8.3	58.3	3.4	70.1
Mill Branch Left Fork	11	5	0.0	25.5	0.0	11.8	75.5	4.8	45.9
Powell River	11	5	0.0	62.7	4.9	9.8	57.8	2.5	57.0
Race Fork UT	16	9	0.0	45.5	2.0	21.8	47.5	4.0	62.7
Roll Pone Branch	11	8	2.5	67.8	0.0	4.2	86.4	2.6	55.3
Spring Branch	16	9	19.0	39.0	7.0	18.0	40.0	3.9	69.7
Cane Branch	8	2	0.0	62.0	0.0	7.4	86.1	2.9	45.9
Kelly Branch	11	6	0.0	58.0	2.0	4.0	68.0	2.8	56.3
Kelly Branch UT	12	5	0.0	16.5	1.9	14.6	76.7	5.2	42.4
Richey Branch	11	6	0.0	51.8	1.8	2.7	86.4	3.3	52.9
Richey Branch UT	15	8	1.0	46.7	8.6	7.6	62.9	3.3	63.0
Spruce Pine Creek	14	8	8.7	24.3	14.8	4.3	69.6	4.5	58.5

Isolating Effects of Total Dissolved Solids on Aquatic Life in Central Appalachian Coalfield Streams.[1]

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Introduction

Background

Elevated levels of total dissolved solids (TDS) have been suggested as stressors to aquatic life in Central Appalachian streams influenced by coal mining (Bodkin et al., 2007, Pond et al., 2008). In coalfield streams, TDS is most often dominated by the dissolved ions SO_4^{2-} and HCO_3^-, with elevated concentrations (relative to reference) of Ca^{2+}, Mg^{2+}, Na^+, K^+, and Cl^- also common (Mount et al., 1997, Pond et al., 2008). At present there are no aquatic life water quality criteria for TDS or dominant ions in the primary coal-producing Central Appalachian states (KY, VA, WV). In all three states, aquatic life conditions are assessed for Clean Water Act compliance using measures of benthic macroinvertebrate community structure.

Dissolved ions, at concentrations above those that occur naturally in Central Appalachian streams, have been shown to cause lethal and sublethal effects to a variety of freshwater invertebrates in laboratory toxicity testing. In mine-influenced streams of the Central Appalachians, in-stream TDS concentration can exceed 2,000 mg/L (Pond et al., 2008). Aquatic bioassays have exposed organisms to a wide range of TDS concentrations. The organisms used in such toxicity testing include common indicator species, such as the cladocerans *Ceriodaphnia dubia* and *Daphnia magna*, the amphipod *Hyalella azteca*, and the midge *Chironomus tentans*, as well as indigenous species such as the mayfly *Isonychia sp*.

Kennedy et al. (2003) exposed *C. dubia* to sulfate-dominated mine effluent and observed significant effects on survival and reproduction at specific conductivities of approximately 6,000 and 3,700 µS/cm (approx. 4,200 & 2,590 mg/L TDS), respectively. Soucek and Kennedy (2005) observed lethal effects of sulfate to *H. azteca* (512 mg/L), *C. dubia* (2,050 mg/L), and *C. tentans* (14,134 mg/L). Chapman et al. (2000) also found reductions in survival and growth of *C. tentans* from simulated mine effluent of approximately 2,000 mg/L TDS. To investigate the effects of TDS on a species representative of populations observed to be impacted at field sites, Kennedy et al. (2004) collected mayflies of the genus *Isonychia* from an unpolluted stream. They then exposed the mayflies to simulated mine effluent for seven days and observed a significant effect on survival at a conductivity of ~1,500 µS/cm (~1,050 mg/L TDS). These laboratory experiments illustrate a clear biological response to elevated TDS, though the results differ among studies, suggesting that TDS tolerance varies widely among different test organisms.

[1] This paper has been previously published as: A.J. Timpano , S.H. Schoenholtz, C.E. Zipper, D.J. Soucek. 2010. Isolating effects of total dissolved solids on aquatic life in central Appalachian coalfield streams. p. 1284 -1302, in: Proceedings, National Meeting of the American Society of Mining and Reclamation.

Additional research has shown that TDS toxicity is dependent upon the type and combination of ions in solution. Acute toxicity tests conducted by Mount et al. (1997) exposed *C. dubia* and *D. magna* to 2,453 different solutions of various ion combinations. They found the relative toxicity of individual ions to be: $K^+ > HCO_3^- \approx Mg^{2+} > Cl^- > SO_4^{2-}$. They also found that toxicity of K^+, Cl^-, and SO_4^{2-} was reduced in solutions with more than one cation present. Soucek and Kennedy (2005) also found that sulfate toxicity was reduced for *C. dubia* and *H. azteca* when water hardness (Ca^{2+} and Mg^{2+}) was increased. Increased chloride concentration also reduced sulfate toxicity to *H. azteca*. These studies suggest that the biological response to in-stream mineral solutions depends on TDS composition and ion interactions, as well as overall ionic strength or TDS concentration.

Field data have shown that the biotic response to elevated TDS also occurs outside the laboratory with indigenous species. Many recent studies of eastern coal mining-influenced streams have found that benthic macroinvertebrate community structure is altered in mining-influenced streams relative to community structure in streams uninfluenced by mining (Green et al., 2000; Pond, 2004; Pond et al., 2008; and others). In those studies, most mining-influenced streams were observed to be elevated in conductivity/TDS and in all cases, one of those water quality parameters has been significantly and strongly correlated with biotic community structure change.

Although field studies have succeeded in demonstrating the ability of benthic macroinvertebrate monitoring to identify aquatic community structural responses to coal mining activity, much remains unknown about how the benthic macroinvertebrate community responds to specific TDS concentrations and compositions. To date, both non-TDS stressors and elevated TDS likely have had concurrent influences on biota in the streams assessed during field studies. Research reported in this paper sought to better characterize the biotic response to elevated TDS and component ions by isolating the TDS variable through the study of streams where non-TDS stressors were minimized.

Objectives

The goals of this research are to better understand how benthic macroinvertebrate community structure responds to elevated TDS where non-TDS stressors are minimized. To that end, we addressed the following questions:

1. Can TDS be reasonably isolated from other stressors in field studies of this type?
2. What is the ionic composition of TDS in headwater streams of Virginia's Central Appalachian coalfield region?
3. How does benthic macroinvertebrate community structure respond to a gradient of TDS/ion concentration?
4. Which measure of water quality – aggregate measures or individual ions - is most related to biotic response?

Methods

Conceptual Approach

This study was designed to quantify how benthic macroinvertebrate community structure responds to various in-stream concentrations of TDS. Given that many factors influence the condition of aquatic life, this study sought to create a single-factor analysis by selecting streams that varied primarily by TDS concentration, while minimizing confounding factors that might influence biota. This was accomplished by seeking study streams with attributes such as habitat quality that are as similar as possible to minimally-disturbed reference streams of the region. The design was intended to ensure that TDS, including its component ions, was the primary factor associated with biotic stress in these streams. Factor-effect levels were studied by examining streams spanning a range of TDS levels. The response factor was benthic macroinvertebrate community structure, which was characterized using several common metrics. Correlation analysis was used to reveal associations between biotic metrics and TDS/ion concentration.

Site Selection

This investigation focused on 1st and 2nd order-, or headwater streams of Virginia's Central Appalachian coalfields. Sharing Omernik Level III Ecoregion 69 and coal-bearing geology with much of eastern Kentucky and southern West Virginia, our study region is comparable to neighboring mining regions (Omernik, 1987).

Candidate site selection was conducted by examining a variety of available data using a GIS, augmented by consultation with mine operators and regulators with specific knowledge of site conditions. Virginia Department of Mines, Minerals and Energy provided data on water quality, mine permits, and historical strip-mining site locations, to which were added aerial photography, landuse data, and firsthand stream knowledge of regulators and mine operators. These data were integrated to develop a list of candidate study sites.

Each candidate site was visited to verify conditions. Physicochemical parameters were measured, with particular interest in pH and conductivity. Site reconnaissance prior to sampling also allowed verification of current land uses and ensured minimal catchment disturbance, as per study design. Physical habitat was evaluated using the qualitative visual estimate approach for high-gradient streams as specified in US EPA's Rapid Bioassessment Protocols (RBP) (Barbour et al., 1999).

The 17 suitable sites selected met non-biological reference criteria commonly used for studies of Virginia non-coastal streams (Burton and Gerritsen, 2003), excepting reference criteria concerning conductivity (Table 1). This was done to keep non-TDS abiotic factors as high-quality and as similar as possible among sites. Test sites meeting these criteria were selected within a range of TDS levels of ~200-2000 mg/L, a range commonly associated with mine-influenced streams of the Central Appalachian coalfields (Pond et al., 2008).

Table 1. Abiotic Criteria for Stream Selection

Parameter or Condition (units or range)	Selection Criterion[1]
Dissolved Oxygen (mg/L)	≥ 6.0
pH (std. units)	≥ 6.0 & ≤ 9.0
Epifaunal substrate score (0-20)[2]	≥ 11
Channel alteration score (0-20)[2]	≥ 11
Sediment deposition score (0-20)[2]	≥ 11
Bank disruptive pressure score (0-20)[2]	≥ 11
Riparian vegetation zone width score, per bank (0-10)[2]	≥ 6
Total RBP habitat score (0-200)[2]	≥ 120
Residential land use immediately upstream	None
Property owner or manager permission for access	Obtained

1 – Parameters and numeric selection criteria from Burton and Gerritsen (2003)

2 – RBP habitat, high gradient streams (Barbour et al., 1999)

In addition to meeting the physicochemical and habitat criteria, all test sites also had to be free from obvious influence from residential land use immediately upstream of the monitoring station. This criterion was important to avoid the unpredictable influence of failing septic systems (*e.g.*, dissolved N, P enrichment) or direct stream discharges of household waste (*e.g.*, particulate organic matter, toxics). Finally, accessibility was a practical criterion that had to be met to allow the site to be included in the study. Access permission was obtained for each study site from private landowners and/or mine permittees.

The goal in choosing reference streams was to identify streams within Virginia's Central Appalachian ecoregion that are as close to minimally disturbed as possible. Streams classified as minimally disturbed represent "…the biological condition in places with a minimal amount of human disturbance." (USEPA, 2006). Although the Virginia coalfield region has an extensive history of human settlement, there are still areas in the region where human impacts are minimal (*e.g.*, National Forests, coal-free geology). To maximize comparability to test sites, reference streams were selected, where possible, that drain coal-bearing geology.

Field Methods

At each study site, benthic macroinvertebrate and water quality samples were collected during the Spring 2009 biological index period (March through May) following the single-habitat approach as specified in Virginia Department of Environmental Quality (VDEQ) Biological Monitoring Program Quality Assurance Project Plan for Wadeable Streams and Rivers (VDEQ, 2008). Approximately 2 m^2 of riffle substrate were sampled using a 0.3 m wide D-frame kicknet with 500 μm mesh. A single composite sample was collected at each site, preserved in 95% ethanol and returned to the laboratory for sorting and identification.

Physicochemical parameters of temperature, dissolved oxygen, specific conductance, and pH were measured *in situ* with a calibrated handheld multi-probe meter (Hydrolab Quanta). Single grab samples of water were collected following modified VDEQ Water Quality Monitoring Standard Operating Procedures (SOP) (VDEQ, 2006). Sampling was modified to use vacuum hand pumps and reusable polyethylene filter assemblies rather than peristaltic pumps and single-use capsule filters. All samples were stored in acid-rinsed polypropylene Nalgene bottles. Samples for dissolved metals, TDS, alkalinity, and major ions were filtered in the field immediately following collection using acid-rinsed cellulose ester filters with a nominal pore size of 0.45μm. Samples for metals analysis were preserved to pH < 2 with 1+1 concentrated nitric acid. All samples were transported on ice and stored at 4 °C prior to analysis in the laboratory. At each site, all biological and water samples were collected concurrently at base flow to minimize alteration of stream chemistry by direct influx of rainwater. All water quality sampling was conducted upstream of and/or immediately prior to biological sampling.

In-stream and riparian habitat quality were assessed at each site to ensure that habitat quality still met criteria (Table 1). Habitat assessment at all sites was performed using the same RBP methods as used during site selection (Barbour et al., 1999).

Laboratory Methods

Biological sample processing followed modified VDEQ Biomonitoring SOP (VDEQ, 2008). Each sample was sub-sampled to obtain a 200 (±10%) organism count following RBP methods (Barbour et al., 1999). Benthic macroinvertebrates were identified to the family/lowest practicable taxonomic level. Most arthropods were identified to family. The groups of Isopoda and Amphipoda (Order), Collembola (Subclass), and Oligochaeta (Subclass) were common exceptions to the family rule. Calculations of biological metrics for each sample were conducted using tolerance values and functional feeding group designations for families (or higher groups as applicable) for Virginia (Burton and Gerritsen, 2003).

An inductively coupled plasma - optical emission spectrometer (Varian Vista MPX ICP-OES w/ICP Expert software) was used to measure dissolved Ca^{2+}, Mg^{2+}, K^+, Na^+, and all species of Cu, Zn, Mn, Se, Al, Fe ions (APHA, 1998). An ion chromatograph (Dionex DX500) was used to measure Cl^- and SO_4^{2-} (APHA, 1998); TDS was measured via filtration of known volumes followed by drying at 180°C (APHA, 1998), with modifications (0.45 micron cellulose ester filter, field filtration); total alkalinity was measured for an aliquot of filtered sample by titration with standard acid (APHA, 1998); and CO_3^{2-}/HCO_3^- were calculated from alkalinity and pH measurements (APHA, 1998).

Data Analysis

All statistical analyses were conducted using JMP 8 (SAS Institute, Inc., Cary, North Carolina). Data were analyzed for correlation among measured water quality variables and biological metrics. Non-normal metrics were natural log transformed prior to analysis where appropriate.

Results & Discussion

Site Selection

The site selection process yielded 17 test streams within Virginia's Central Appalachian coalfield region where elevated TDS was the primary stressor to aquatic life (Figure 1; Timpano et al., 2009). These streams were very similar to reference conditions in all other respects.

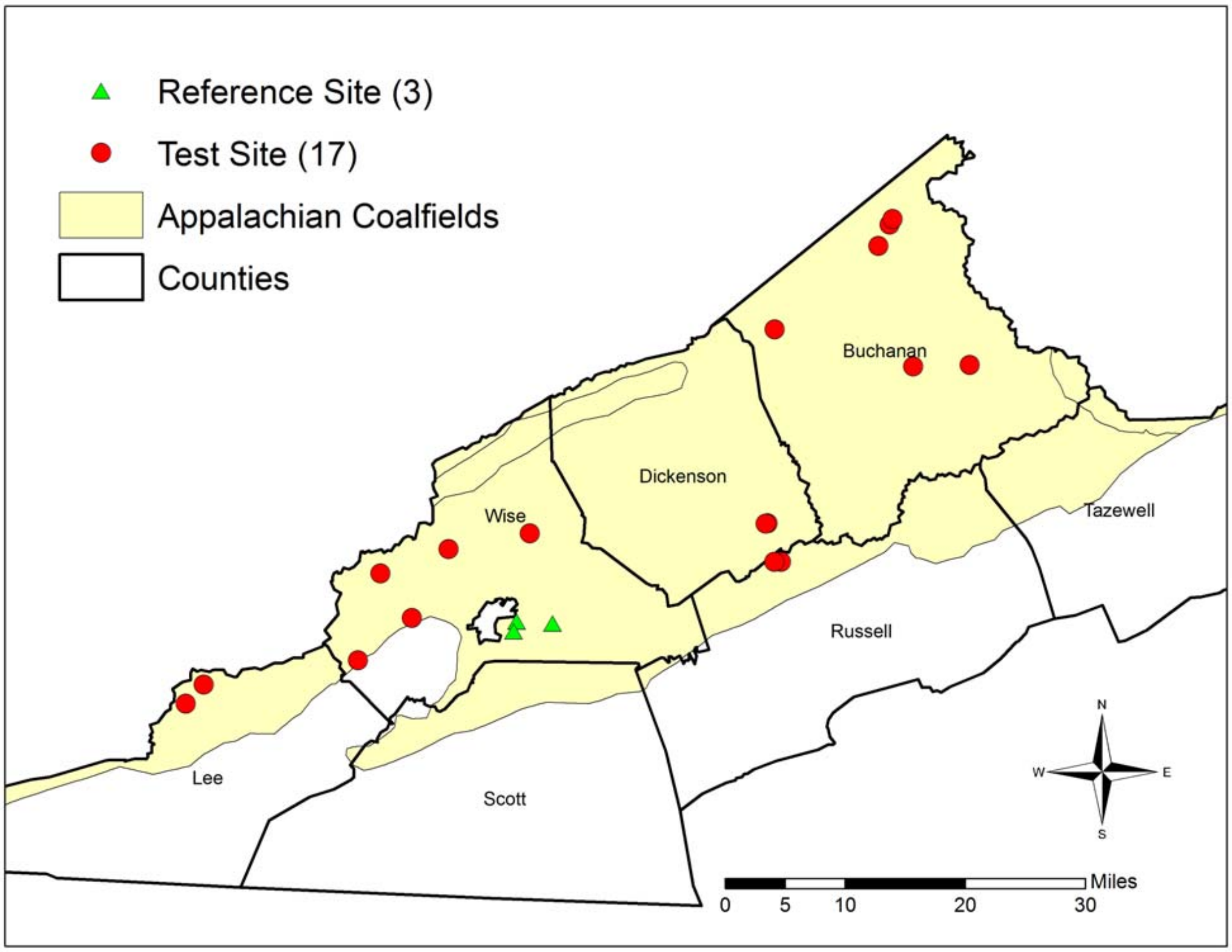

Figure 1. Reference and test sites in the Central Appalachian coalfields of Virginia.

Reference-quality streams draining coal-bearing geology are rare in Virginia's Central Appalachian ecoregion. With an extensive history of mining and associated development, nearly all of the region's coal-geology streams are influenced by a combination of legacy strip-mining, contemporary mining, infrastructure, commercial, industrial, and/or residential development. Three reference streams were located in the Jefferson National Forest in the southern portion of the coalfields based on recommendations from VDEQ biologists, because these streams had served previously as regional references during VDEQ special studies (Figure 1; Timpano et al., 2009). Data from reference streams were used to establish reference-level habitat quality to ensure that test site habitat was comparable to reference.

Habitat scores for each of the 17 test sites were compared to the mean habitat score of the three reference sites and classified based on comparability to reference. All test sites scored >85% of reference, corresponding to a rating of "Comparable to Reference" (Barbour et al., 1999). Such comparability was deemed acceptable in effectively minimizing the influence of habitat quality on biotic condition, thus allowing the effects of TDS to be isolated and measured.

Physicochemical criteria for pH and dissolved oxygen were within reference limits for all sites. Trace metal analysis indicated water column dissolved metal concentrations below method detection limits for nearly all samples, with no measurements above chronic criteria for copper, aluminum, zinc, or iron.

Water Chemistry

Test sites exhibited a range of TDS/ion concentrations (Table 2). Among test sites, SO_4^{2-} was the most common ion by weight, followed by HCO_3^- and Ca^{2+} (Figure 2). Ion composition generally conformed to that pattern across test sites (Figure 3).

Table 2. Distribution statistics for selected water quality parameters (test sites only, n=17). All units mg/L unless noted.

	Temp (°C)	pH (SU)	DO (mg/L)	Cond. (µS/cm)	TDS	SO_4^{2-}	HCO_3^-	Ca^{2+}	Mg^{2+}	Na^+	K^+	Cl^-
					- - - - - -	- - - - - -	- - - - - -	mg/L	- - - - - -	- - - - - -	- - - - - -	- - - - - -
Minimum	10.9	6.6	7.8	25	27.8	4.2	5.1	2.2	1.5	1.0	1.4	0.9
10th %tile	12.1	7.1	8.2	216	126.2	56.6	19.6	22.5	10.9	4.3	1.8	1.0
Median	13.4	7.9	9.3	490	298.0	155.0	89.9	45.2	24.0	18.5	2.9	1.9
90th %tile	15.0	8.3	10.0	856	593.1	420.5	173.2	95.7	54.8	52.2	4.4	8.6
Maximum	17.5	8.5	10.2	970	791.6	531.4	301.7	119.9	75.4	135.9	5.0	9.8

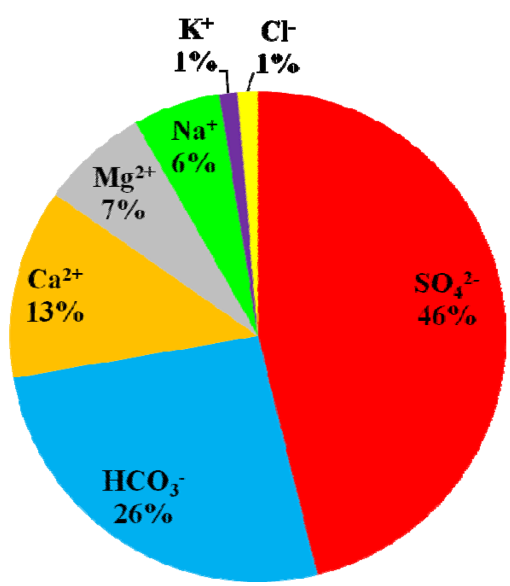

Figure 2. Mean relative proportions, by weight, of major ions (test sites only, n=17).

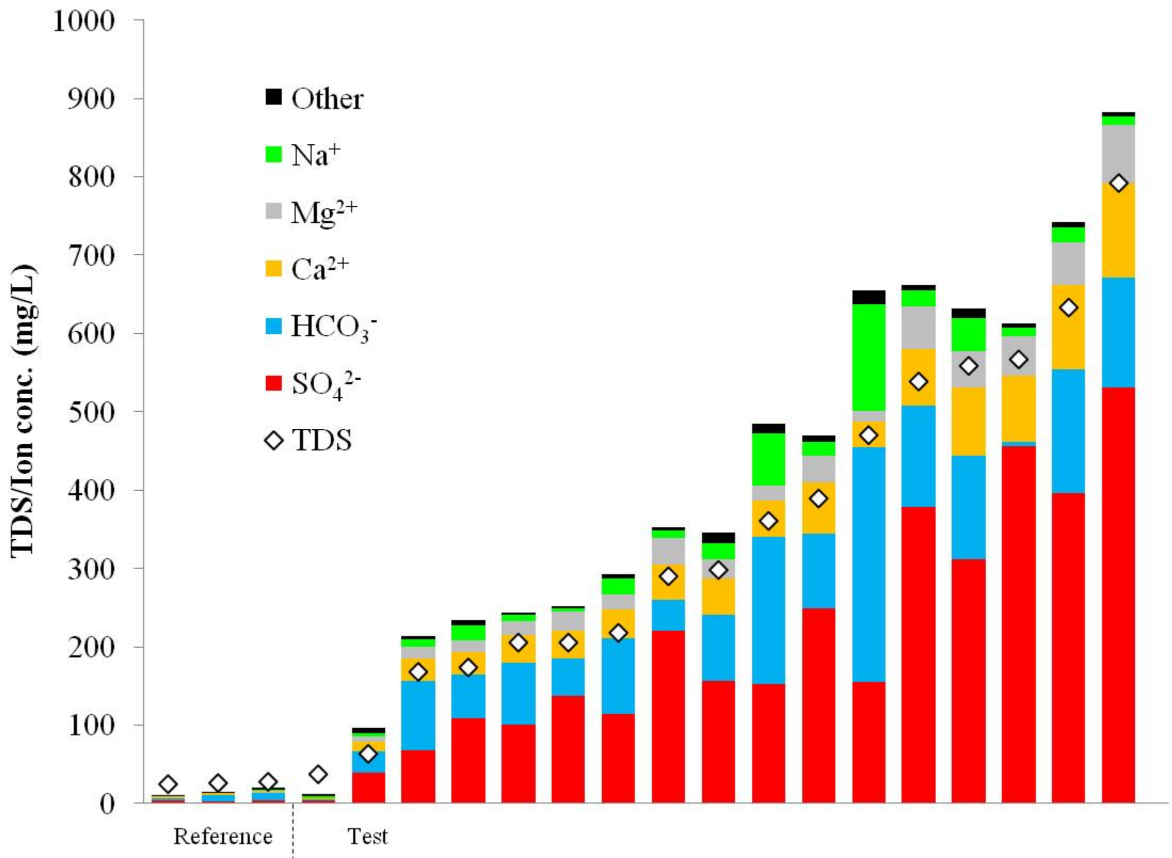

Figure 3. TDS and major component ion concentrations for all study sites.

Water quality parameters were highly correlated (Table 3), suggesting that a single parameter such as conductivity, TDS, or sulfate could represent water quality for a site.

Table 3. Pearson product-moment correlations for major ions and related measures (test sites only, n=17). All correlations shown are significant ($p < 0.05$).

	Ca^{2+}	SO_4^{2-}	TDS	Cond.	Mg^{2+}	K^+
SO_4^{2-}	0.96					
TDS	0.94	0.94				
Cond.	0.92	0.90	0.98			
Mg^{2+}	0.96	0.98	0.92	0.87		
K^+	0.76	0.70	0.85	0.87	0.69	
HCO_3^-			0.50	0.56		0.68

Associations Between Biology and Water Quality

Many metrics of benthic macroinvertebrate community structure and function were considered (Table 4).

Table 4. Candidate biological metrics by category, with expected response to anthropogenic disturbances such as water quality degradation; and metric definition; superscripts designate transformation used for statistical analysis (if applicable). Adapted from Burton and Gerritsen (2003).

Metric by Category	Response	Definition
Taxonomic Richness		*Number of different taxa in each specified group*
Number of Taxa	decrease	Total number of different taxa in sample
Number of EPT Taxa	decrease	Number of different taxa in the generally sensitive orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)
Number of E Taxa	decrease	Number of different Ephemeroptera taxa (mayflies)
Number of P Taxa	decrease	Number of different Plecoptera taxa (stoneflies)
Number of T Taxa	decrease	Number of different Trichoptera taxa (caddisflies)
Number of Diptera Taxa	decrease	Number of different Diptera taxa ("true" flies, such as midges and blackflies)
Composition		*Percent of individuals within total sample of…*
Percent EPT	decrease	…the generally sensitive orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)
Percent E	decrease	…the order Ephemeroptera (mayflies)
Percent Plecoptera	decrease	…the order Plecoptera (stoneflies)
Percent Trichoptera[b]	decrease	…the order Trichoptera (caddisflies)
Percent EPT less Hydropsychidae	decrease	…the generally sensitive orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), less the generally tolerant Trichoptera family Hydropsychidae
Percent PT Less Hyd.	decrease	…the generally sensitive orders Plecoptera (stoneflies) and Trichoptera (caddisflies), less the generally tolerant Trichoptera family Hydropsychidae
Percent T less Hydropsychidae [b]	decrease	…the generally sensitive order Trichoptera (caddisflies), less the generally tolerant Trichoptera family Hydropsychidae
Percent Diptera[a]	increase	…the order Diptera ("true" flies, such as midges and blackflies)
Percent Chironomidae[a]	increase	…the Diptera family Chironomidae (midges)
Percent Baetidae	variable	…the generally tolerant Ephemeroptera family Baetidae
Percent Nemouridae	variable	…the generally tolerant Plecoptera family Nemouridae
Percent Hydropsychidae [b]	variable	…the generally tolerant Trichoptera family Hydropsychidae

Table 4, continued. Candidate biological metrics by category, with expected response to perturbation, definition, and transformation used (if applicable). Adapted from Burton and Gerritsen (2003).

Metric by Category	Response	Definition
Diversity		*Percent of individuals in sample that comprise…*
Percent 1 Dominant Taxon	increase	…the most abundant taxon
Percent 2 Dominant Taxa	increase	…the two most abundant taxa
Percent 5 Dominant Taxa	increase	…the five most abundant taxa
Trophic Groups		*Percent of individuals in sample (or number of different taxa), that obtain food by…*
Percent Collectors[a]	decrease	…collecting/gathering depositional organic matter
Percent Filterers[b]	variable	…filtering suspended organic matter
Percent Predators	variable	…preying on other organisms
Percent Scrapers	decrease	…scraping algae and associated material from substrate surfaces
Percent Shredders	decrease	…shredding/chewing coarse organic matter such as leaves and detritus
Number of Collector Taxa	decrease	(number of taxa classified primarily as Collectors)
Number of Filterer Taxa	variable	(number of taxa classified primarily as Filterers)
Number of Predator Taxa	variable	(number of taxa classified primarily as Predators)
Number of Scraper Taxa	decrease	(number of taxa classified primarily as Scrapers)
Number of Shredder Taxa	decrease	(number of taxa classified primarily as Shredders)
Tolerance		
Hilsenhoff Biotic Index	increase	abundance-weighted average of organic pollution tolerance for assemblage

[a] transformed to normal distribution for analysis as Ln(X)
[b] transformed to normal distribution for analysis as Ln(1+X)

Correlation analysis was conducted on the variables, either as measured or as transformed to normal distributions. Significant correlations were noted (Table 5). All relationships between water quality parameters and biotic metrics were negative or not significant (Figure 4).

Table 5. Pearson product-moment correlations for biological metrics and water quality parameters (test sites only, n=17). Metrics are as defined in Burton and Gerritsen (2003). All correlations are significant ($p < 0.05$).

Metric	Ca^{2+}	SO_4^{2-}	Mg^{2+}	TDS	Cond.	K^+
Number of EPT Taxa	-0.81	-0.81	-0.79	-0.76	-0.76	-0.64
Number of E Taxa	-0.75	-0.79	-0.77	-0.71	-0.71	-0.59
Number of P Taxa	-0.78	-0.75	-0.73	-0.72	-0.72	-0.60
Percent 5 Dominant Taxa	0.75	0.71	0.71	0.64	0.62	0.52
Number of Collector Taxa	-0.66	-0.71	-0.71	-0.61	-0.58	-0.55
Number of Taxa	-0.63	-0.56	-0.57	-0.50	-0.49	
Number of T Taxa	-0.52					

No single water quality parameter stood out as a lone predictor of biological condition, but SO_4^{2-} may be the best choice among the water quality parameters we measured if use of a single parameter is desired. Correlation analysis revealed that Ca^{2+}, SO_4^{2-}, and Mg^{2+} are significantly correlated to the greatest number of biological metrics, each more so than TDS or conductivity, whereas HCO_3^- and Na^+ were not significantly correlated with any biological metrics. Sulfate was the dominant ion in nearly all samples, whereas Ca^{2+} and Mg^{2+} contribute less to TDS (Figures 2 & 3). In addition, observed maximum SO_4^{2-} concentration (531 mg/L) was much higher than maximum concentrations of Ca^{2+} (120 mg/L) or Mg^{2+} (75 mg/L) (Table 2). Furthermore, the three ions are all strongly correlated to each other (Table 3). Sulfate has been used as a reliable indicator of mining activity (Pond et al., 2008). These data suggest that SO_4^{2-} may be a suitable candidate for prediction of aquatic life response in mining-influenced streams, although that preliminary finding will be subject to additional investigation by further study.

The strongest correlations to TDS and related measures were with the generally sensitive groups of aquatic insects (the orders Ephemeroptera, Plecoptera, and Trichoptera, or EPT). In this regard, our results echo the findings of other studies of mining-influenced streams that also observed correlations of elevated TDS with reduced richness of EPT taxa (*e.g.*, Pond et al., 2008; Pond, 2004; Green et al., 2000).

We did not observe the Percent Ephemeroptera (mayflies) metric to be correlated with any chemical variables, including TDS (Figure 4). Other studies of mining-influenced streams have observed significant negative correlations between mayfly relative abundance and TDS or conductivity (*e.g.*, Pond et al., 2008; Pond, 2004; Green et al., 2000). However, while mayfly relative abundance was not correlated with dissolved ion measures in our data, mayfly richness (Number of E Taxa) was negatively correlated with TDS and related measures (Table 5). This result demonstrates that at our study sites for the Spring 2009 sampling period, as TDS increased across sites, the benthic macroinvertebrate community structure responded by shifting to fewer mayfly taxa, but overall mayfly relative abundance did not respond predictably to increasing TDS (Figure 4).

The metrics most strongly correlated with TDS and related parameters were measures of community richness, rather than measures of relative abundance. This suggests that although family-level community richness (the number of taxa present) may decline with increasing TDS, abundance of individuals within the remaining families, and perhaps overall order abundance, may remain less affected, at least within the range of TDS and component ion concentrations that we assessed.

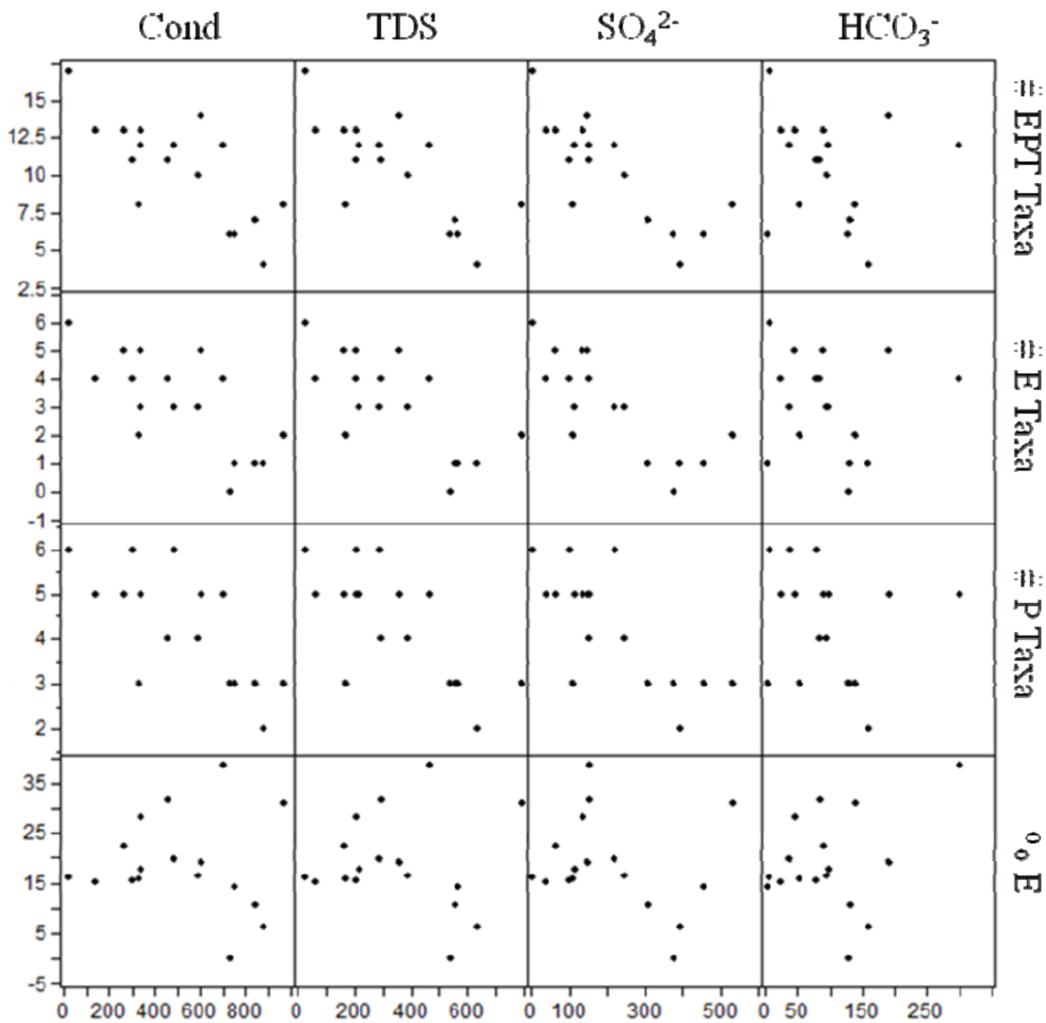

Figure 4. Scatter plot matrix of select chemical and biological parameters (test sites only, n=17). Units are mg/L except conductivity (µS/cm). All correlations are significant ($p < 0.05$) except those involving bicarbonate (HCO_3^-) and Percent Ephemeroptera (% E).

Conclusions

We found that TDS can be reasonably isolated from other stressors in the Virginia Central Appalachian coalfields through strategic site selection, as we were successful in locating sites with elevated TDS that exhibited minimal influence from non-TDS stressors. This was critical to understanding the influence of dissolved ions on benthic macroinvertebrate community structure with minimal confounding from common TDS covariates.

The TDS in waters we sampled tended to be dominated by SO_4^{2-} and HCO_3^-, with Ca^{2+} being the third most prevalent ion by mass. Bicarbonate was not significantly correlated with any benthic community metrics. Sulfate comprised nearly half of the dissolved ion mass on average, and it was the dominant water quality parameter most highly correlated with biotic metrics. For these reasons, we think SO_4^{2-} concentration (w/v) is a candidate for use as a single-parameter predictor of biological condition.

Biological metrics that exhibited a significant negative correlation with TDS/ions were those of family-level community richness, especially those measuring richness of the generally sensitive insect orders of Ephemeroptera, Plecoptera, and Trichoptera. Order-level relative abundance metrics, including Percent Ephemeroptera, were not correlated with conductivity, TDS, or component ion concentration. This suggests that while family-level community richness (the number of taxa present) may decline with increasing TDS, abundance of individuals within the remaining families, and perhaps overall order abundance, may remain less affected, at least within the range of TDS and component ion concentrations that we assessed.

Acknowledgements

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Long-Term Mine Soil Weathering and TDS Release: Do Topsoil Substitutes Really Mimic Natural Soils?

2009/2010 Powell River Project Annual Progress Report

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Introduction and Background

The Surface Mine Control and Reclamation Act (SMCRA) of 1977 contained a number of contentious provisions including return to original contour (AOC), long-term liability bonding periods, and return to "equal or better" post-mining land use conditions. However, one of the more stealthy provisions was SMCRA's allowance for use of pre-selected overburden materials as topsoil substitutes when (A) the native A+E horizon materials are less than 6 inches thick, and (B) the physical and chemical properties of the proposed substitute spoil materials are deemed suitable for such use. Since native topsoil layers throughout the Appalachian coalfields are usually less than six inches thick, and removing them from steep slopes is difficult and expensive, the vast majority of coal mined lands in the region have employed topsoil substitutes.

In 1982, the USDI Office of Surface Mining and the Powell River Project co-funded the installation of the Controlled Overburden Placement (COP) experiment to objectively assess the viability of the topsoil substitute concept and to determine whether or not organic amendments would be beneficial. In one component of the COP experiment we are directly comparing five mixes of sandstone:siltstone (SS:SiS) overburden while in a separate experiment we are following the effects of topsoil return, sawdust addition and four incremental loading rates of biosolids. All treatments are replicated four times and the plots are split between herbaceous (dominantly tall fescue) and forest (red oak following pine) vegetation. We intensively monitored those two side-by-side experiments through the late 1980's, and our results can be reviewed at the PRP web site and at http://www.cses.vt.edu/revegetation/minereclam.html. In summary, we found that (A) properly selected and placed spoil materials provided an outstanding soil medium for tall fescue production and allowed vigorous invasion of native herbaceous species; (B) higher pH spoils such as the siltstone strata employed were deleterious to pine tree growth; and (C) higher rates of biosolids amendments drove high fescue production while suppressing the pines. The COP experiment remains the longest intact and continuously monitored study of mine soil genesis in the World. Follow-up studies by our group at other sites in the 1990's and early 2000's also characterized the wider effects of biosolids applications and the nature of inherent variability in mine soil properties in the Research & Education Center. However, very little detailed soil analyses have ever been performed on the native pre-mining soils in the Research & Education Center area for direct comparison.

Over the past decade, the concept of topsoil substitution has been directly and indirectly criticized from a number of perspectives. First of all, advocates of the return of Appalachian mined lands to native forest covers have pointed to the lack of topsoil salvage and the inclusion of higher pH unweathered spoils as directly inhibiting effective reforestation. These objections have been raised by citizens and certain well-trained scientists alike. Secondly, the fact that relatively unweathered spoils (such as those employed in the COP study) release significant total dissolved solids (TDS) loads to drainage waters over time has been implicated as a component of mining related surface water degradation under both low and moderate pH conditions. In fact, it now appears that mining discharges will be directly regulated for TDS (or by proxy by EC – electrical conductance) over time and reducing bulk TDS may be a much more difficult water treatment proposition for the coal industry than limiting more conventional parameters such as total Fe and Mn. Finally, the ability of these mine soils to accumulate organic matter, maintain a stable and viable microbial biomass and available nutrient pools, and overall productivity potentials beyond the requisite five-year performance liability period is also questioned by many citizens' groups.

In 2007, we proposed to directly address a number of these challenges by initiating a new program of mine soil sampling and analysis utilizing our established baseline experiments at the Research and Education Center, and at other locations where long-term baseline data sets are available, that will allow us to study changes in mine soil properties and productivity relationships over prolonged periods of time. Furthermore, we will directly compare mine soil properties for a range of important parameters (e.g. pH, organic matter content, P-forms, microbial biomass) with a suite of unmined native soils forming out of the same rocks. Thus, by a combination of direct and differential analysis, we originally proposed to meet the following objectives:

Research Objectives

1. To determine the long-term (20+ years) effects of overburden rock type and surface treatments on important mine soil morphological, physical, chemical and microbiological properties.

2. To directly compare the properties of weathering mine soils of varying age with unmined native soils formed from the same strata.

3. To measure the net TDS elution potential of a range of fresh, partially weathered and well-weathered topsoil substitute materials.

4. To predict the ability of selected overburden materials to weather and transform into mine soils suitable for the support of native hardwoods and hayland/pasture vegetation, and to estimate the rate of transformation.

Methods and Procedures

Overall Approach

We are fortunate to have an array of well-characterized, documented and "preserved" research sites throughout the Powell River Project Research & Education Center area and the surrounding region. These include the COP experiment, areas to the north of Powell River that have been minimally disturbed since 1990, and certain limited locations south of Powell River that have not been re-mined since 1990. While much of the 1990 aged mine soil surface received uniform treatment of biosolids+compost, there are significant areas of that surface that did not. By differentially sampling across these contrasting treatment areas, we will be able to directly determine the net effect of organic matter additions on long term soil development process and important mine soil productivity parameters.

Furthermore, the recent re-mining activity to the south of Powell River will allow us to sample and "pair up" mine soil pedons that are very young (1 to 10 years) with much older mine soils (25+ years) to the north that formed out of identical parent materials. Finally, we also have access to a range of relatively intact native forest soils in the overall Powell River area that occur between mining disturbances.

We are now completing the third year of this study. In year one, we focused field work on collecting a wide range of unweathered and weathered spoil types in the region and on sampling pedons within the immediate vicinity of the Research & Education Center as described above. In year two, we worked with Jim Burger and other collaborators to locate additional pedons where we can be assured of good "control" of spoil age, type, and treatments, and where we have access to archived original spoil samples or original data sets to determine rates of change of various mine soil properties. In the laboratory we focused on characterizing the chemical and physical properties of these soils, as well as on initiating column leaching studies to characterize the potential leaching behavior of various mine spoil materials pH, EC/TDS, and cations and anions related to acid-base reactions (Ca, Fe, HCO_3^-, and SO_4^{2-}). In the third year, we focused on completion of the column leaching studies. As discussed later, we still have several study components that we intend to finish over the fall and winter of 2010/2011 which include the description, sampling and analysis of the 28 year-old mine soils in the COP experiment and several more natural forest soils from the Wise County area.

Progress to Date (August 2010)

In year one of this study fifteen samples representing fresh, partially weathered and well-weathered topsoil substitute materials were collected from PRP and other mines in southwest Virginia and east Kentucky. These samples represent a variety of spoil types including sandstone, siltstone and shale in different proportions and at various degrees of weathering. Some geologic, chemical and physical characteristics for ten spoil samples used in the leaching studies are presented in Tables 1 and 2. During year two, an eight month leaching column study allowed characterization of element release from three spoil materials including sandstone (OSM 1), a mudstone (OSM 2), and a mix of materials (OSM 11). The leaching columns were built from PVC pipe with a diameter of 7.6 cm and a length of 40 cm. The samples were run in triplicate under saturated and unsaturated conditions (6 columns per sample), and were leached and sampled twice a week using a simulated rainfall solution (pH 4.8). In year three, the columns were re-established with nine additional spoil samples (OSM 3, OSM 4, OSM 6, OSM 7, OSM 10, OSM 12, OSM 14, COP SS, COP SiS) to more completely represent the various spoil types at various degrees of weathering. The second leaching column study lasted one year, and the leachate samples were analyzed for pH, electrical conductivity (EC), Al. Ca, Fe, K, Mg, Mn, Na and S.

Table 1. Geologic description and associated information for 10 mine spoils used in column leaching studies.

Lab-ID	Material	Geologic Description	Geologic formation	Coal Seam	Particle size %	
					< 1 cm	> 1 cm
OSM 1	Mine spoil Unweathered	98% unweathered, gray and orange, medium to coarse grained, feldspathic sandstone; 2% unweathered gray silty mudstone. (No coal apparent.)	Norton	Raven1	23	77
OSM 2	Mine spoil Unweathered	93% dark gray carbonaceous silty mudstone; 6% unweathered, gray, fine-grained sandstone/siltstone; 1% coal.	Four Corners Formation; Breathitt Group	Hazard #7 and Hazard #8	60	40
OSM 3	Mine spoil Partially-Weathered	50% highly weathered, gray and orange, fine grained and medium to coarse grained, feldspathic sandstone; 30% unweathered gray silty mudstone ; 10% unweathered feldspathic sandstone; 8% unweathered gray silty mudstone; 2% coal.	Middle Wise	Kelly/Imboden	87	13
OSM 4	Mine spoil Weathered	98% weathered, reddish-brown silty mudstone; 1% weathered sandstone; 1% coal.	Lower Wise	Clintwood/Blair	85	15
OSM 6	Mine spoil Unweathered	90% minimally weathered gray clayey siltstone; 10% brown silty mudstone; trace coal.	Lower Wise	Clintwood/Blair	79	21
OSM 7	Mine spoil Weathered	85% weathered brown-gray silty mudstone; 13% unweathered gray silty mudstone; 2% weathered sandstone. (No coal apparent.)	Middle Wise	Kelly/Imboden	62	38
OSM 10	Mine spoil Unweathered	97% unweathered gray silty mudstone; 2% coal; <1% unweathered sandstone; <1% brown shale.	Upper-middle Wise	Phillips	72	28
OSM 11	Mine spoil Weathered	99% weathered sandstone; 1% silty mudstone; trace coal.	Upper-middle Wise	Taggart	68	32
OSM 12	Mine spoil Unweathered	98% unweathered, gray, medium grained sandstone; 1% weathered sandstone; 1% silty mudstone.	Upper-middle Wise	Taggart	45	55
OSM 14	Mine spoil Weathered	80% weathered, gray and orange, feldspathic sandstone; 20% gray silty mudstone. (No coal apparent.)	Lower Wise	Clintwood/Blair	65	35

Table 2. Selected chemical properties of the 10 mine spoils used in column leaching studies.

	2:1	Saturated paste		PPA[1]	CCE[2]	Total S
Lab-ID	**pH**	**pH**	**EC dS m^{-1}**	**Tons CCE / 1000 Tons**	**%**	%
OSM 1	7.98	6.88	1.27	0	2.7	0.06
OSM 2	7.14	7.04	3.48	0	4.6	0.23
OSM 3	5.24	6.93	0.94	3.58	1.3	0.07
OSM 4	5.22	6.46	0.29	0.22	1.6	0.03
OSM 6	7.02	7.26	1.40	0	2.1	0.14
OSM 7	7.03	7.66	0.20	0.12	4.7	0.03
OSM 10	8.14	7.85	0.66	0	6.0	0.11
OSM 11	5.24	6.28	0.56	0.28	3.7	0.03
OSM 12	8.64	7.84	0.40	0	5.3	0.12
OSM 14	5.74	7.49	0.36	0.12	3.4	0.03

[1] Potential Peroxide Acidity (PPA). Values shown represent net acidity/lime demand.
[2] Calcium Carbonate Equivalent (CCE)

Characterization of Chemical Properties

The saturated paste pH (Table 2) of these samples was generally in the neutral to alkaline range which is typical of fresh, relatively unweathered materials from this region due to hydrolysis reactions involving broken primary mineral grains and carbonate dissolution. The soluble salt content, indicated by EC (Table 2), produced by most fresh spoils was relatively low (< 1.0 dS m^{-1} or 1000 µS/cm), although a few values ranged up to 3.5 dS m^{-1}. Elevated EC values appeared somewhat related to total S and/or CCE (Table 2). While these values are quite typical for soil:water systems, any mine site effluent/leaching levels in excess of 0.5 dS m^{-1} (500 µS/cm) would pose significant regulatory concern. Total S content was relatively low, ranging up to 0.23%, which is not unusual for the region sampled. Correspondingly, PPA values (Table 2) were low (< 3.6 tons $CaCO_3$/1000 tons material) due to low S content in combination with moderate CCE values.

A comparison of the pH values determined by saturated paste (pH_{sp}) versus in a 2:1 water:sample mix ($pH_{2:1}$; Table 2) indicated lower pH values by the 2:1 method for the samples at $pH_{2:1}$ < 7.0. This may be due to the difference in equilibration time (much longer for the saturated paste method) allowing alkaline mineral surfaces to react with acidic components. As expected, weathered mine spoils tended towards lower pH. Only one weathered sample (OSM 7) was neutral ($pH_{2:1}$ = 7.03), while the other weathered samples had $pH_{2:1}$ values between 5.22 and 5.74. Among the mixed partially weathered/unweathered mine spoils those with higher percentages of weathered material generated lower pH.

Leaching Column Trial

The data presented in Figures 1-11 report results from 45 leaching events over the initial 22 week period. The data are presented in groupings to contrast the results from weathered versus unweathered materials and saturated versus unsaturated conditions. We highlight pH, EC, sulfate, bicarbonate and other elements particularly relevant to acid-base reactions. All data presented below represent the mean observations from three replicate columns. Detail on column replicability (which was outstanding) can be found in Daniels et al. (2009).

pH: For most materials used in this study, leachate pH (Figs 1-2) from the first leaching events was substantially lower than the initial saturated paste pH data on the fresh bulk materials (pH_b). This difference was more readily observed in weathered than in unweathered materials, and was least apparent for unweathered sandstone. With few exceptions, leachate pH increased over the first few leaching events, and achieved a relatively stable equilibrium within 10 to 20 leaching events. The unsaturated columns were usually higher in pH than for the same material when saturated. Under saturated conditions, only samples OSM 1, OSM 2, and OSM 11 equilibrated to pH values near pH_b, while the other saturated samples maintained pH values that were approximately 0.3 to 1.0 pH unit less than pH_b. The higher pH levels from unsaturated conditions could be due to the effects of CO_2 partial pressures on carbonate dissolution in the unsaturated columns or perhaps siderite ($FeCO_3$) formation in the saturated columns.

Electrical Conductance (EC): Analyses completed on the first set of columns indicated a high correlation coefficient (r = 0.98) between EC and TDS, as expected, indicating that EC can be used as an effective proxy for TDS (Daniels et al. 2009). Leachate EC from unweathered mine spoil (Fig. 3-4) was consistently higher than from partially oxidized and weathered samples of similar geology. This difference was more pronounced for mudstone than for sandstone. For most samples, EC values dropped quickly achieving a steady state within 10 to 20 leaching events and maintained relatively low levels for the remainder of the leaching trial. However, OSM 2 and 6, both unweathered mudstones, maintained values greater >1.0 dS m^{-1} through the first 10 leachings. These high EC values may indicate a highly reactive sulfide phase (framboidal?) that while present in relatively low amounts (0.23% and 0.14% S), reacted quickly with substrate carbonates to produce prolonged sulfate release.

Bicarbonate: Bicarbonate analysis was completed only on the first set of columns which included OSM 1, OSM 2, and OSM 11. The release of bicarbonate (Fig. 5) reflected the effects of saturated versus unsaturated conditions more than any other measured parameter in this study with vastly more bicarbonate released under saturated conditions. The lower levels from the unsaturated columns presumably reflect acid neutralization reactions, even when relatively low amounts of S were present.

Sulfate: Sulfate release patterns (Fig. 6-7) reflect the acid-base reactions discussed above which are due to fundamental differences in the geology/mineralogy of these materials such as total-S content (Table 2), degree of "pre-oxidation" via weathering and trace carbonate content. In SW Virginia, the majority of strata within the Pennsylvanian system are low in pyritic-S. Many of the massive sandstones that dominate the Lee, Norton, and Wise formations contain secondary carbonate cementing agents (Howard et al., 1988) which offset the relatively minor amounts of sulfidic minerals found in most geologic sections. Significant accumulations of sulfides do occur in coal seams and underclays throughout the region; however, these seams are relatively thin (< 3 m). Also, several relatively minor sections of overburden in Virginia (e.g. the Standiford seam interburden of the middle Wise formation) generate spoils with significant (>20 Mg/1000 Mg) levels of potential acidity (Orndorff & Daniels, 2004). For most spoil samples used in this study

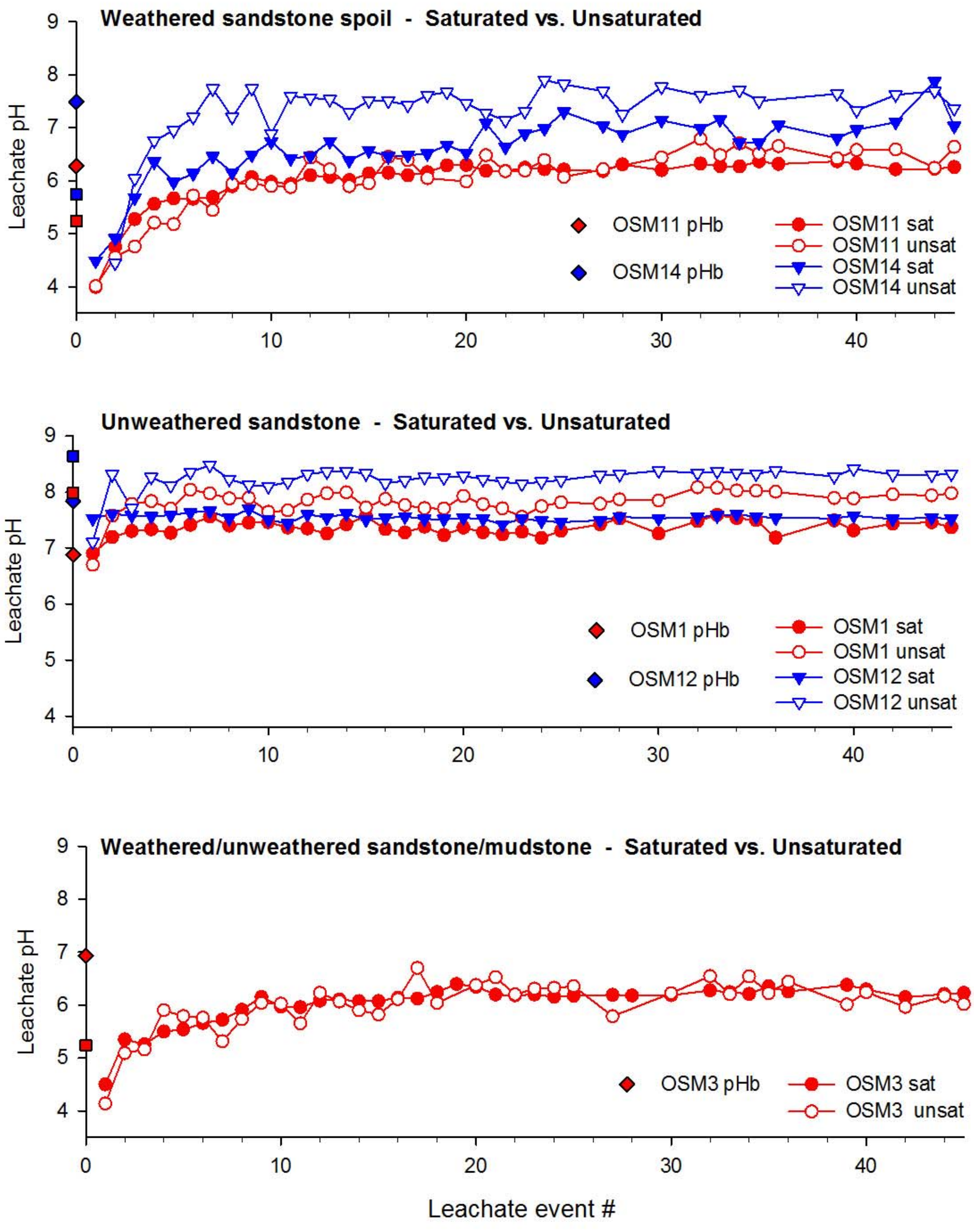

Figure 1. Leachate pH from weathered and unweathered sandstone spoil and mixed spoil material under saturated and unsaturated conditions. Saturated paste pH of the initial bulk material (pH_b) is indicated by symbols on the Y axis. The 45 leaching events occurred over 22 weeks.

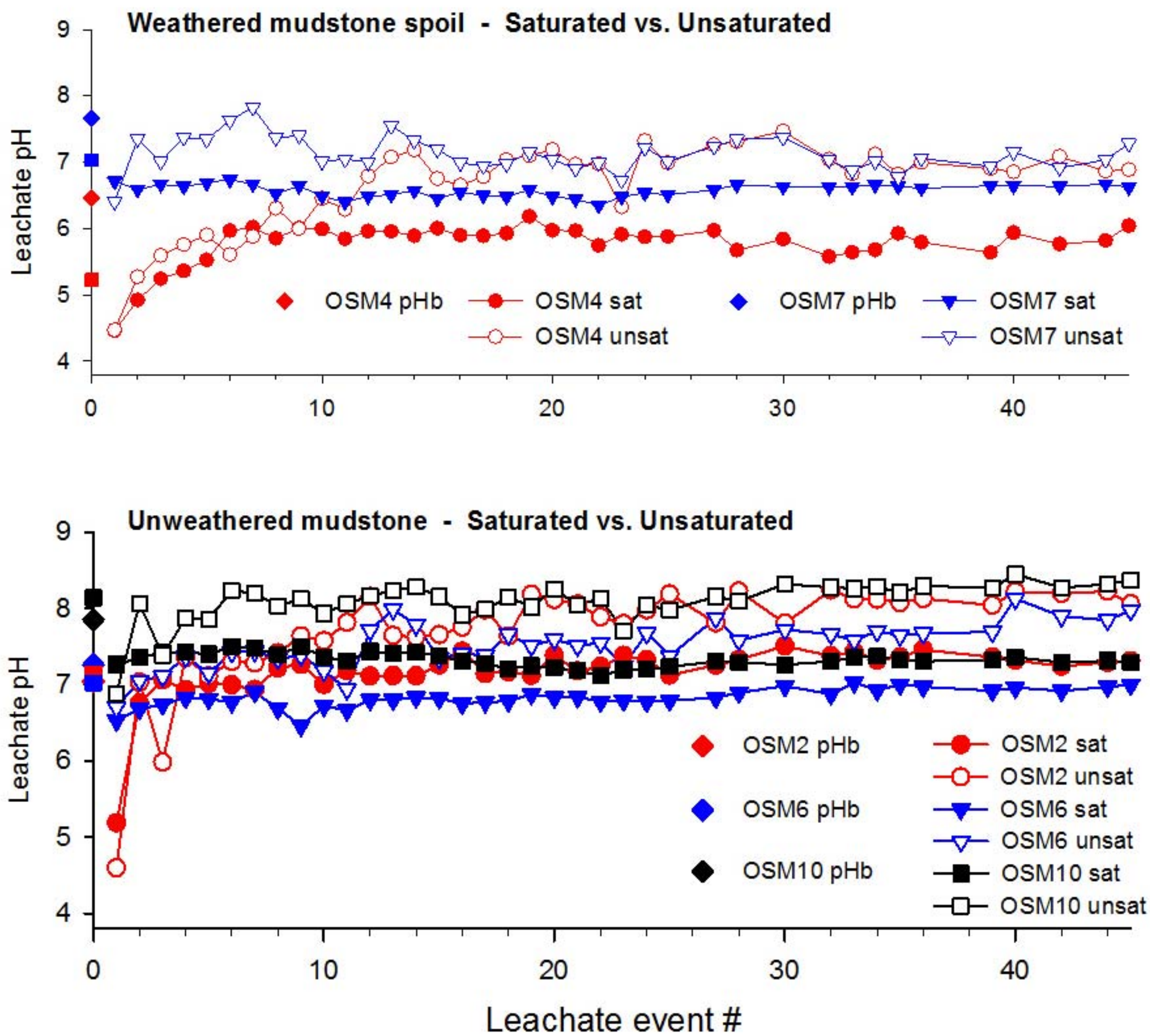

Figure 2. Leachate pH from weathered and unweathered mudstone spoils under saturated and unsaturated conditions. Saturated paste pH of the initial bulk material (pH_b) is indicated by symbols on the Y axis. The 45 leaching events occurred over 22 weeks.

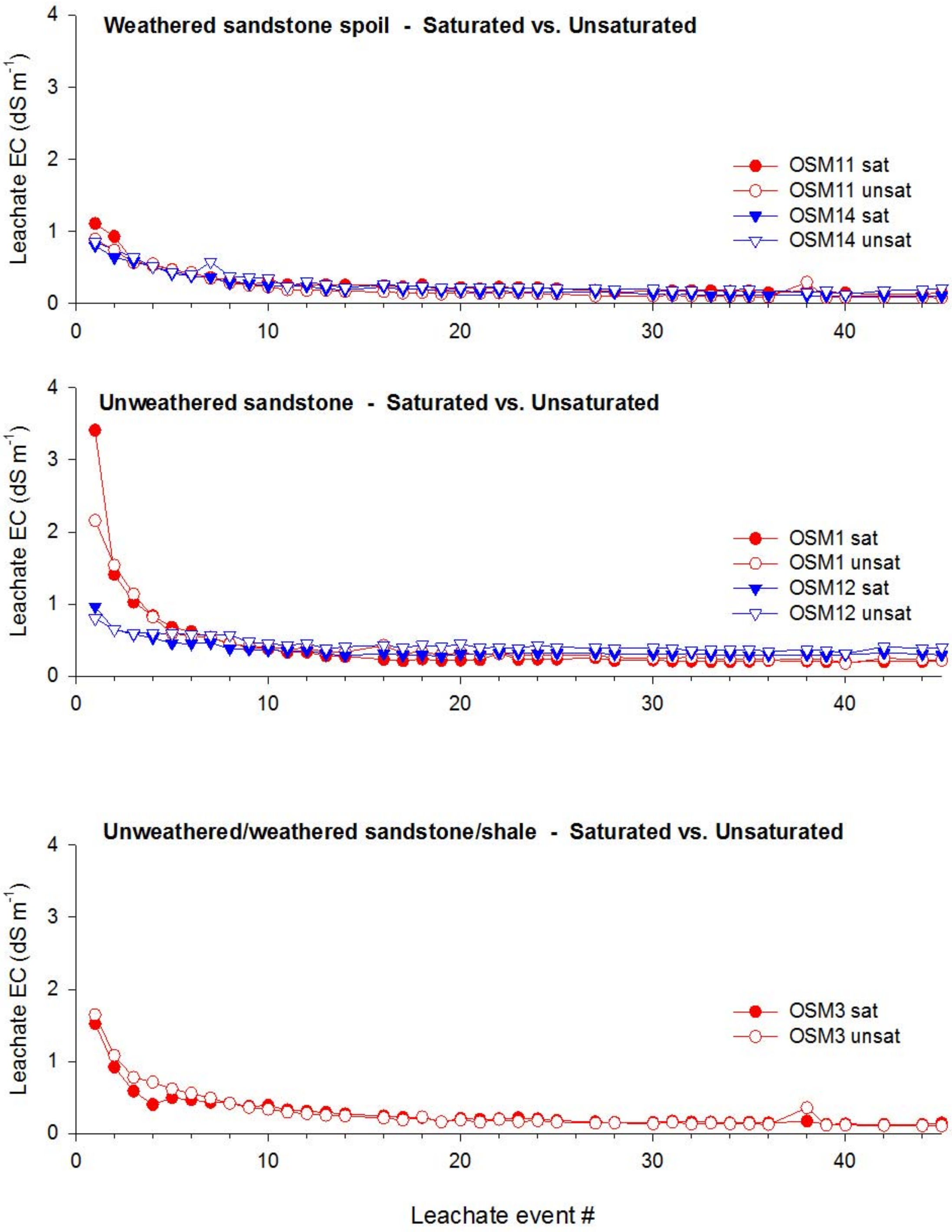

Figure 3. Leachate EC from weathered and unweathered sandstone spoil, and mixed spoil material, under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks.

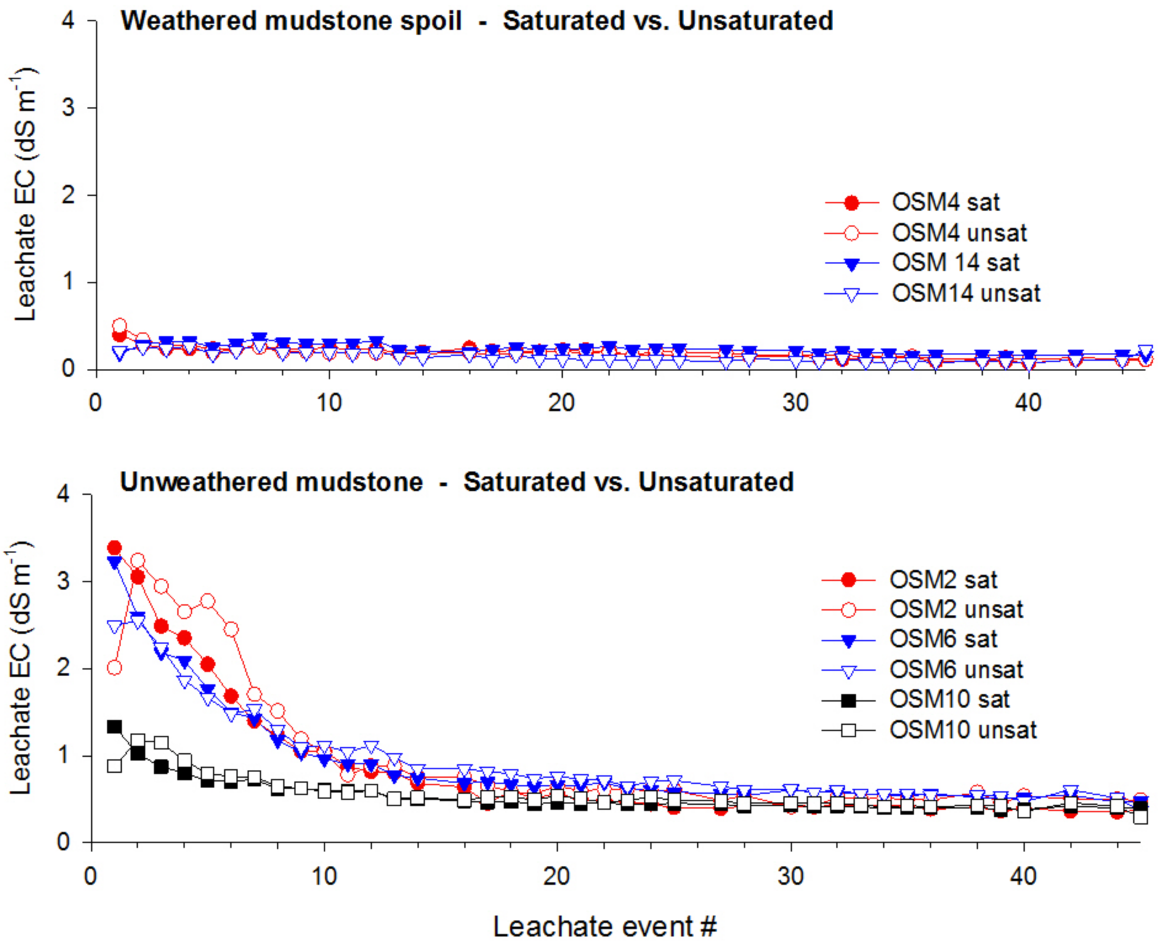

Figure 4. Leachate EC from weathered and unweathered mudstone spoil under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks.

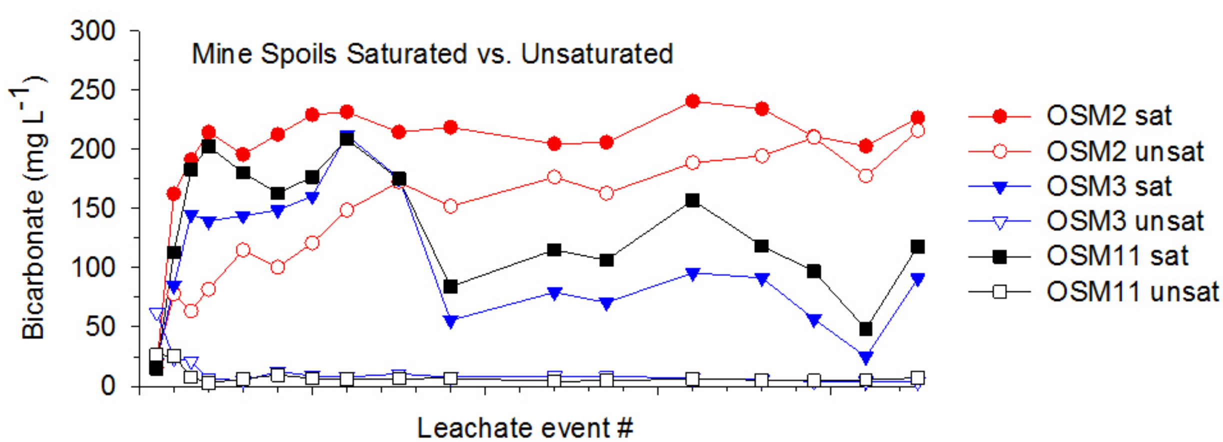

Figure 5. Bicarbonate leached from selected mine spoils under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks.

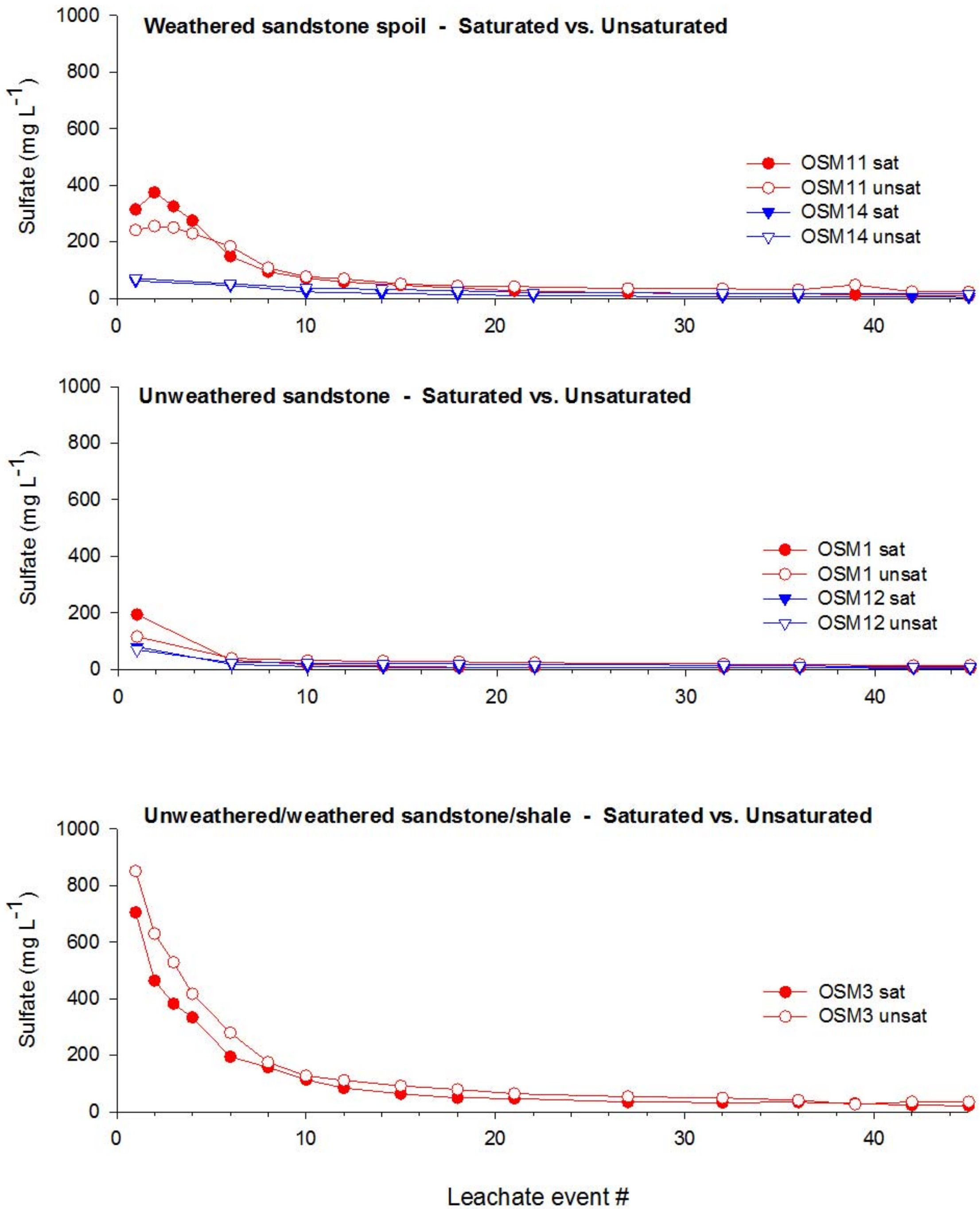

Figure 6. Sulfate leached from weathered and unweathered sandstone, and mixed spoil material, under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks. Note: for Sulfate graphs Y-axis ranges from 0 – 1000 mg L^{-1} for Fig. 6 and 0 – 3000 mg L^{-1} for Fig. 7 (below).

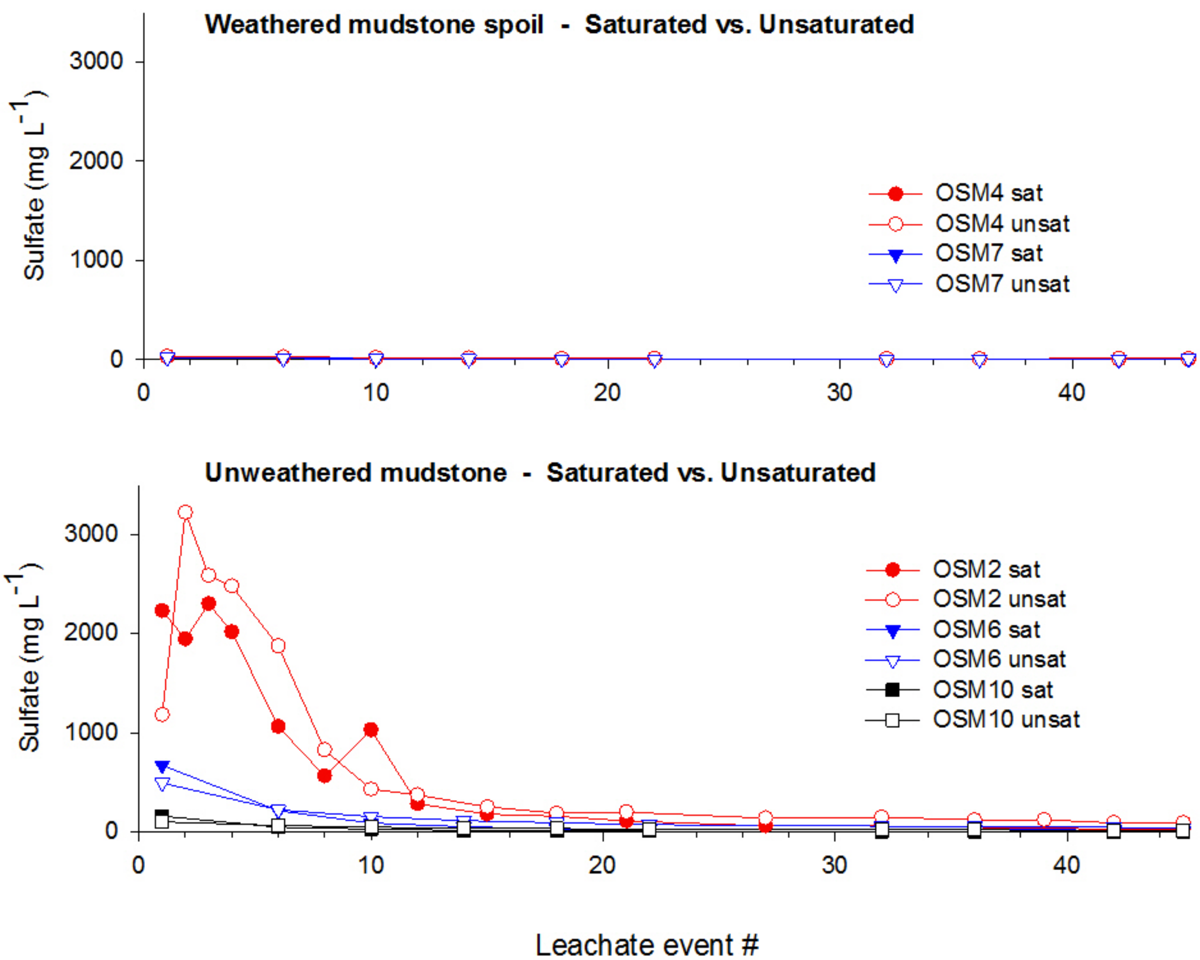

Figure 7. Sulfate leached from weathered and unweathered mudstone under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks. Note: for Sulfate graphs Y-axis ranges from 0 – 1000 mg L^{-1} for Fig. 6 (above) and 0 – 3000 mg L^{-1} for Fig. 7.

the majority of sulfate release occurred within the first month (10 leaching events or over approximately one pore volume of total elution). The greatest sulfate release occurred from OSM 2, a mudstone from the Breathitt Formation (eastern Kentucky) which has been noted and studied for its acid forming potential (Barnhisel and Massey, 1969).

<u>Calcium (Ca), Iron (Fe) and Manganese (Mn).</u> Calcium leaching patterns (Figs. 8-9) for the mine spoils followed the pattern of sulfate very closely. As expected, unweathered spoil materials released greater amounts of Ca than did the weathered materials. Iron release patterns (Fig. 10) did not mirror other pyrite oxidation reaction products such as sulfate. Most samples released low concentrations of Fe over the course of the study. Due to the relatively high pH of the leachates, the vast majority of Fe was precipitated and retained within the columns in various

oxy-hydroxide forms. The relatively high release of Fe from one saturated weathered mudstone spoil (OSM4) and one saturated weathered sandstone spoil (OSM 11) is notable and may be the result of reduction of original Fe-oxides to soluble Fe^{+2} over time.

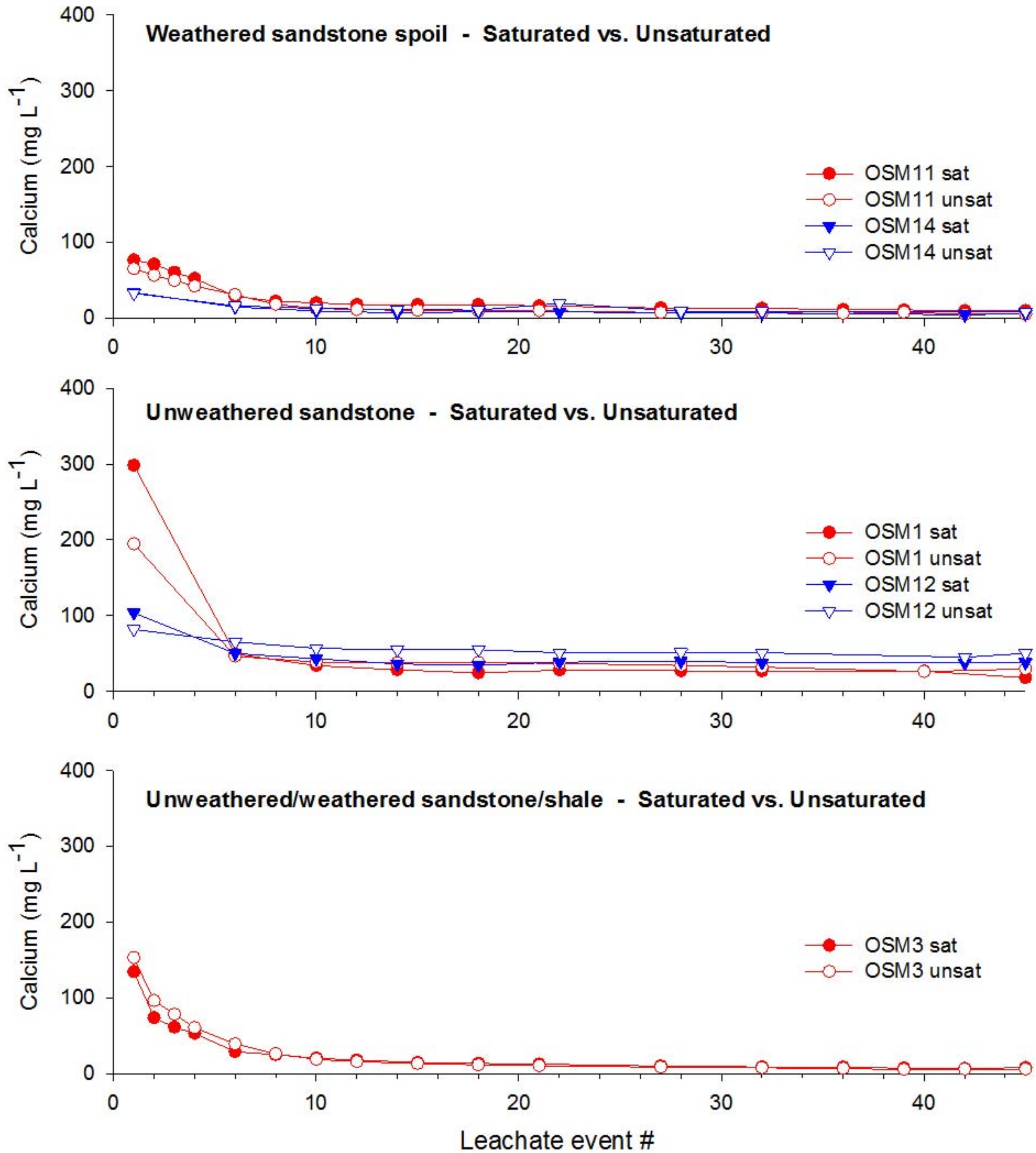

Figure 8. Leachate Ca from weathered and unweathered sandstone spoils under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks. Note: for Ca graphs Y-axis ranges from 0 – 400 mg L^{-1} for Fig. 8 and 0 – 800 mg L^{-1} for Fig. 9 (below).

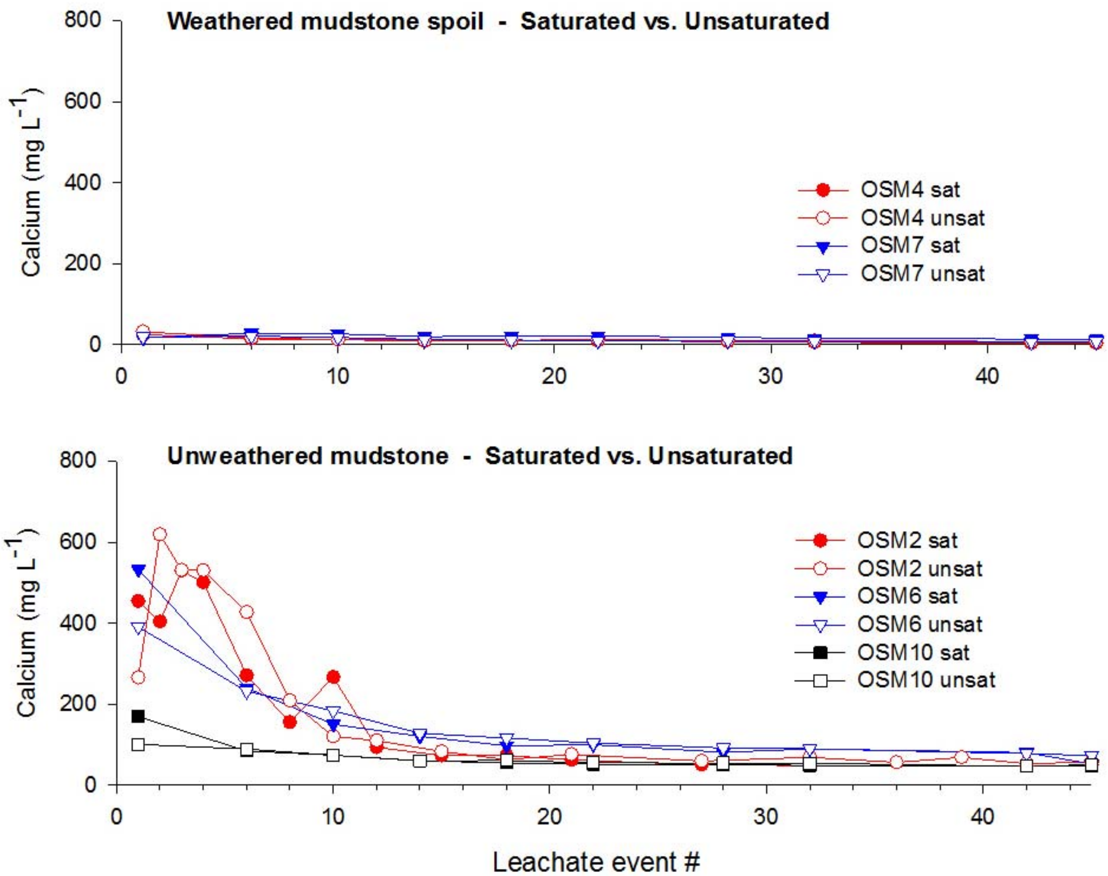

Figure 9. Leachate Ca from selected weathered and unweathered mudstone under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks. Note: for Ca graphs Y-axis ranges from 0 – 400 mg L^{-1} for Fig. 8 (above) and 0 – 800 mg L^{-1} for Figs 9.

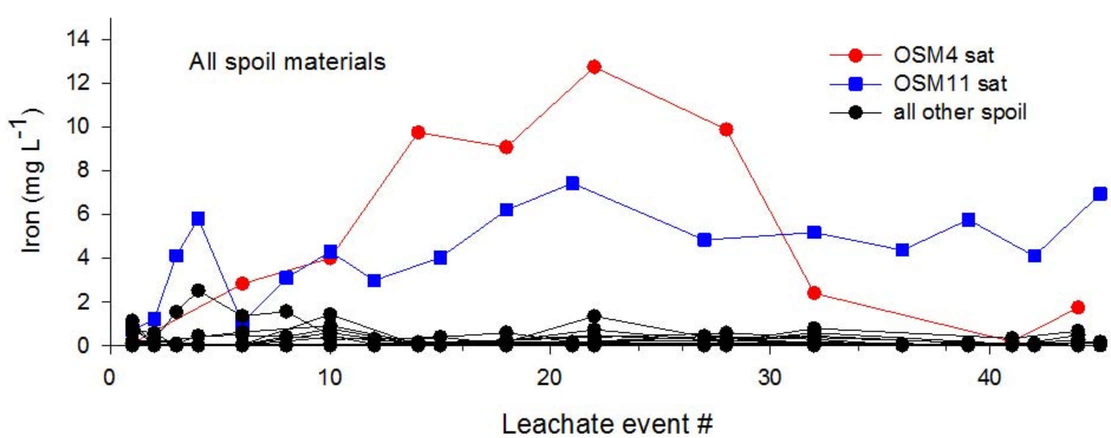

Figure 10. Leachate Fe from all spoil materials under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks.

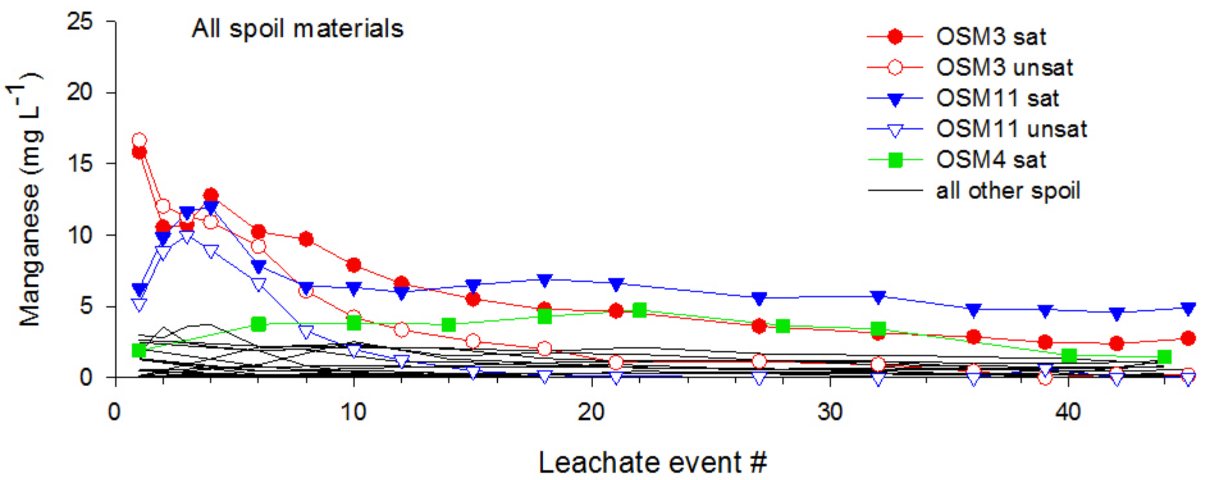

Figure 11. Leachate Mn from all spoil materials under saturated and unsaturated conditions. The 45 leaching events occurred over 22 weeks.

Leachate Mn concentrations typically were higher than Fe due to the higher solubility of Mn than Fe in this pH range (Fig. 11). Furthermore, Evangelou (1995) pointed out that poorly crystalline Mn^{+4} oxides may serve as an alternative electron acceptor (oxidizing agent) in moderate pH sulfide oxidation environments which can lead to large amounts of water soluble Mn^{+2} being mobilized. For some samples, particularly OSM 1 and OSM 3, Mn release patterns were closely related to Ca and sulfate, however, the opposite was true for the Kentucky shale (OSM 2). This is presumably due to differences in the inherent Mn oxide forms in the differing spoils. For both Mn and Fe, the most notable releases occurred under saturated conditions.

Data Analysis, Synthesis and Expected Results

Our combined results from the mine spoil characterization and column leaching components of this program clearly point out that differences in (A) mine spoil strata/origin and associated mineralogy interact very strongly with (B) the extent of *in situ* pre-mining weathering to profoundly influence the bulk chemical properties of resultant mine soils and their potential to generate TDS to contact waters (e.g. leachates). Thus, it is clear that the mining industry should isolate and utilize oxidized and pre-weathered (e.g. brown) materials whenever possible for their topsoil substitutes if their long term management goal is establishment of forest vegetation. This practice will also result in a minimization of TDS release to surface runoff waters. Our results indicate that many partially or non-weathered (e.g. gray) spoils in our region have the potential to produce substantial leachate TDS levels, particularly over their initial leaching cycles. However, our data also clearly indicate that these elevated leachate levels will drop dramatically over time to levels below current regulatory concern (e.g. 500 µS/cm) for most materials. At this point in time the critical question seems to be: *Just how long will it take for TDS levels to drop in a field setting*? We believe that it should be possible to predict TDS elution behavior for a given spoil material based upon a combination of analyses such as routine acid base accounting, hydrogen peroxide potential acidity, saturated paste EC, and other relatively quick lab procedures. These rapid tests will need to be "calibrated" against column data, however, and then appropriate scaling factors will need to be developed to relate the output results to expected field conditions to answer the critical question posed above.

Collectively, it is also clear from our data set that TDS elution is driven by fundamental acid-base reactions in the spoils (e.g. pyrite oxidation coupled with carbonate dissolution/ neutralization) and that these leachates will commonly be circumneutral in pH and relatively low in Fe, Mn and other metals of concern. Spoil leachate TDS levels are driven principally by acid-base reactions products, primarily sulfate and calcium. Thus, when large volumes of these materials are placed into valley fills and highwall backfills and allowed to freely interact with infiltrating water, significant TDS elution should be expected. Alternative fill designs that limit water infiltration into (and percolation through) these materials need to be developed and tested.

To date, we have completed approximately 85% of our originally proposed work plan, but we still need to complete the sampling and analysis of the COP experiment and several more natural and fully weathered forest soils from the SW Virginia region that correlate with our bulk spoil sampling events from 2007. This delay was due to the fact that it became very clear to us by mid-2009 that we needed to focus our attention on TDS elution prediction. However, we do intend to complete this work over the fall/winter of 2010/2011 as an in-kind contribution to the project. Once this final aspect of our work is completed, we will be able to directly determine and report the relative effect of rock type and surface treatments in the COP experiment on 25+ years of mixed herbaceous vegetation and tree growth. Similarly, by comparing the properties of the biosolids treated and untreated areas of nearby 15 year-old Taggart mine soils, we will be able to confirm overall rates of important mine soil transformation such as pH reduction and organic matter accumulation in an initially high pH sandstone system. By then comparing the bulk salt and acid extractable nutrient+metal data for each pedon with depth, we will be able to estimate the mass “TDS leaching potential” of each mine spoil material and assess how much of the TDS load appears to have leached over 15 to 25+ year time spans and from what depth. These data and findings will be reinforced by our spoil leaching column trials. Finally, we will directly compare and contrast all mine soil pedons with nearby natural soils over the same strata.

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Select carbon dynamics as functional indicators of restoration success: a research approach

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Executive Summary

Permitting of coal mining in the central Appalachian region is dependent on maintaining or restoring hydrologic and ecological function in streams affected by coal extraction. As mandated by the Clean Water Act (CWA) [section 404; compensatory mitigation rule], mining operations permitted by the U.S. Army Corps of Engineers (USACE) must mitigate "unavoidable" stream impacts attributable to valley fill and other mining activities (Federal Register April 10, 2008). Assessment of stream ecosystem structure and function is fundamental to determining ecological condition and success of mitigation techniques employed on these streams. Traditional bioassessment techniques for freshwater streams are cost-effective, efficient, and are often conceptually linked to ecosystem processes through the characterization of functional feeding groups (FFGs) within the benthic macroinvertebrate assemblage. Although common, reliance on measures of biotic assemblage structure may limit the utility of these indices as reliable surrogates for stream function. In contrast, direct measures of ecosystem processes are rarely implemented in the context of restoration assessment.

Gessner and Chauvet (2002) identified lack of expertise, as well as temporal and monetary constraints as primary arguments against using direct, process-based measures as indicators of functional condition. Objections to direct functional assessment persist because monitoring entities require that measures of ecological condition be: (1) easily implemented in space and time, (2) objective, (3) causally linked to stream processes or assemblage structure, (4) sensitive to changes in environmental quality, and (5) able to separate anthropogenic disturbance signals from natural environmental stochasticity. Carbon (C) as organic matter serves as an essential habitat and energy compartment within headwater and downstream environments. Thus, assessment of carbon dynamics is integral to determining levels of ecosystem function for streams receiving restoration treatments.

This project will assess the functional status of eight low-order streams affected by coal mining that have received restoration practices and four reference streams using measures that are relatively simple to implement and relate directly to stream carbon dynamics. Background data collection and physical characterization of these streams are already underway, and a research plan has been developed to assess the functional condition of these streams. Based on the selected measures of C dynamics, this research will:

1. assess the condition of coalfield streams receiving restoration practices relative to forested reference streams,
2. examine relationships between functional and structural assessments of reconstructed streams and reference streams,
3. determine factors affecting these processes in streams receiving restoration practices,
4. evaluate these measures as indicators of stream condition.

Background and Justification:

Stream Restoration Assessment. In the context of ecological restoration of streams, Kauffman et al. (1997) identified the need for reestablishment of biotic and physiochemical factors associated with stream ecosystem function. Though river and stream restoration practices are increasingly common, follow-up assessment of restoration success is generally lacking (Bernhardt et al. 2005; Bernhardt et al. 2007). Post-restoration monitoring is required in mined-land streams as a condition of CWA [section 404] permits. Until recently, assessment of these streams has focused on evaluation of ecosystem structure via measures of water chemistry, channel stability, and biotic assemblages. These practices are likely to change, however, as recent guidance from USEPA (2010) states that quantitative, functional stream impact assessment should be used to determine environmental effects of mining and the capacity of mitigation efforts to restore functions.

Important Carbon Functions: Stream ecology studies have shown headwater streams to be predominantly open ecosystems, governed by allochthonous inputs of organic matter (e.g., Fisher and Likens 1973; Cummins 1974; Vannote et al. 1980; Hall et al. 2000). Organic matter from upstream and riparian zones serves as both energy source and habitat substrate for stream biota and, as such, is an essential component of ecosystem function. Though studies of C budgets provide a holistic view of ecosystem energy flow via measurement of storage, inputs, and outputs, these studies require high-intensity sampling efforts and are costly, thus limiting their utility within management and assessment frameworks. With regard to assessment of C dynamics, measures of organic matter transport, production, and detrital processing within stream corridors provide manageable alternatives to assessment of comprehensive C budgets. These parameters provide insight into the energetic function of stream ecosystems, while restricting sampling requirements to manageable levels. In conjunction with structural assessment and routine physiochemical monitoring, assessment of C dynamics within restored streams not only provides a sound basis for determining levels of ecosystem production and biotic activity, but also allows for insight into factors that control these processes. The C functions to be assessed in this study are:

- *Riparian organic matter subsidies.* The quality and quantity of organic matter input to and transported through the stream corridor greatly influence biotic assemblage structure which, in turn, influences the quality and quantity of organic matter available to downstream environments. Riparian inputs of coarse particulate organic matter (CPOM) as leaves or twigs and branches, for instance, may undergo leaching or fractionation to produce dissolved organic matter (DOM) or fine particulate organic matter (FPOM), respectively. The magnitude and rate of these processes depends on spatiotemporally heterogeneous factors, such as season and stream type, as well as the quality and quantity of CPOM entering the stream across ecosystem boundaries. As such, characterization of terrestrial subsidies to the aquatic system as leaves and other forms of litter is an essential component to determining the availability of resources to the biotic assemblage with clear implications for downstream environments. Ideally, comprehensive understanding of CPOM inputs would include contributions from upstream of research sites as well as the riparian zone. However, in-stream sample nets require frequent retrieval to keep from filling and may interfere with concurrent experiments. In contrast direct-fall inputs of litter can be readily characterized over relatively long periods, while still providing information about resources available to *in situ* and downstream assemblages.

- *Leaf litter processing.* Gessner and Chauvet (2002) have presented a compelling case for use of leaf litter breakdown in functional assessments of stream ecosystems. Moreover, multiple studies have identified leaf litter and organic matter processing as ecosystem functions essential to maintenance of *in situ* and downstream environments (Wallace et al. 1982a; Wallace et al. 1982b; Fisher and Gray 1983; Hutchens and Wallace 2002; Simmons et al. 2008; Aldridge et al. 2009; Benstead et al. 2009). Leaf litter breakdown is a function of both biotic processes, such as microbial and macroinvertebrate activity, and abiotic processes, such as chemical breakdown and leaching of organic compounds. Because biotic activity may be indexed by leaf-litter breakdown rates (Simmons et al. 2008), breakdown coefficients may be regarded as bioindicators of functional condition. Leaf litter breakdown rates integrate changes in environmental quality over time. Comparison of leaf litter breakdown rates of reaches that have received restoration practices to those of reference and/or pre-restoration conditions allows for quantitative and objective assessment of ecosystem function and subsequently can serve as an indicator of ecological restoration success.

- *Primary Production.* *In situ* rates of primary production may be determined by changes in dissolved O_2 or CO_2 concentration, pH, ^{14}C incorporation, or indexed by the change in standing crop over time, given control for losses caused by grazing, scour, and migration (Steinman et al. 2006). Although each method has advantages and limitations, efficient assessment is predicated upon constraints of time and funding. Chlorophyll *a* (chl *a*) is the predominant photopigment common to all primary producers, and concentrations thereof have been used as surrogates for algal standing crop. Measurement of chl *a* is not a demographic measure of population or community (i.e., biotic structure), and as such, calculated differences in this photopigment over time can be used to estimate accrual rates of algal assemblages. Accrual rates incorporate gains in the number and size of individuals, as well as any losses due to herbivory, scour, or sloughing. Conceptually, accrual rates of benthic algae and net primary production (NPP) should be roughly equivalent when controlled for losses due to physical and biotic processes.

Progress to date:

Site Selection. To date, >100 watercourses were evaluated for potential inclusion in this study. A preliminary assessment of ecosystem functions for 6 restored/reconstructed streams and 3 non-restored/reconstructed streams was conducted by previous members of the research team (Northington et al. 2009), and we have continued to collect physiochemical and biotic samples on these streams.

Additional streams have been identified to augment the population of streams used in the previous study. Reconstructed/restored streams affected by active deep mine discharge, as well as those deemed incomparable based on physical characteristics were excluded from the study. Minimally impacted reference streams were identified as those that lack evidence of recent significant watershed disturbance by humans, and with specific conductance $\leq$ 150 µS cm^{-1} and circumneutral pH (6-8), Eight reconstructed streams and four reference streams were finally selected for study (Table 1).

Table 1. Location and general description of streams selected for study.

Stream Name	Location	Stream Type[1]	Stream Order[2]
Sewing Creek	Buchanan County, VA	MRR	1
Shooting Range Creek	Buchanan County, VA	MRR	1
Chaney Creek	Russel County, VA	MRR	2
Laurel Branch	Russel County, VA	MRR	1
Stonecoal Creek	Russel County, VA	MRR	1
Callahan Creek	Wise County, VA	MRR	1
Critical Fork	Wise County, VA	MRR	2
Guest Mountain #3	Wise County, VA	MRR	2
Copperhead Branch	Buchanan County, VA	UFR	1
Big Branch	Dickenson County, VA	UFR	1
Crooked Branch	Dickenson County, VA	UFR	2
Middle Camp Branch	Dickenson County, VA	UFR	1

[1] Two stream type categories have been identified, mined receiving restoration practices (MRR) and un-mined forested reference (UFR).
[2] Stream order was determined using 7.5' USGS quadrangles.

The majority of eight reconstructed streams selected have received restoration treatments within the past five years. Primarily because of variation in techniques, these stream corridors are characterized by riparian zones in multiple stages of development. Relatively few restoration efforts located are >5 years old, however, and those with developing forest in the watershed generally lack mature riparian canopy. In an effort to expand the study, additional contacts were made to assist in locating mining-impacted streams where restoration or reconstruction efforts have resulted in establishment of more developed riparian canopy. Additionally, "comparably canopied" reference streams, which are relatively unimpacted by other disturbance were sought to discern between scale-dependent (e.g., local riparian vs. watershed forest) controls on selected C dynamics. No such streams were discovered after several scouting trips and contacts, and as such, we are confident that streams of this nature are rare in the region, if they exist.

Experimental Design. The twelve streams selected are within the Central Appalachian ecoregion (69; level III). These first- and second order perennial and intermittent streams are contained within the Appalachian plateau physiographic region of Virginia. Sixty m stream reaches of contiguous channel morphology and riparian structure were delineated and subdivided into two reaches of 30 m for different components of the study. Reaches have been classified into two major categories: un-mined forested riparian (UFR), and mined receiving restoration efforts (MRR). It must be noted that the last class, MRR, is comprised of streams receiving a wide variety of treatments from channel construction alone to those receiving Natural Channel Design (NCD) efforts and diverse riparian plantings. Site history, reconstruction method, temporal estimates of mining and stream restoration occurrence, and physical descriptions of the watercourse are being documented for each stream, and more qualitative information will be collected with the aid of contacts already made.

Materials and Methods.

Habitat and Structure

Physiochemical. Temperature data loggers (HOBO U22-001 Water Temp Pro v2; Onset Computer Corp.) will be deployed in September 2010 and will record stream temperature at 2-h intervals through September 2012. Units will be deployed within 0.5 m of litter bags (see below). Sampling of selected water chemistry variables will begin September 2010, continuing at bi-monthly intervals through July 2012. Dissolved oxygen (mg L^{-1}), temperature (°C), specific conductance (µS cm^{-1}), and pH will be measured in a uniformly mixed portion of the streamwater column immediately below the surface using a Hydrolab water quality meter (Hydrolab Quanta, Hach Instruments, Loveland, CO) at all sites for each sampling date. Additionally, grab samples will be collected at all sites for water chemistry determinations and treated according to Standard Methods (APHA 2005). Following filtration through Millipore brand mixed cellulose ester filters (47 mm diameter, 0.45 µm pore size) samples will be either preserved in 1+1 HNO_3 for determination of heavy metals and major cations, or transported on ice for determination of alkalinity, total dissolved solids (TDS), ammonium (NH_4^+-N), total oxidized nitrogen (NO_3^-N+ NO_2^-N), and soluble reactive phosphate (PO_4^{-3}-P). Flow-injection analysis will be used to determine ammonium (NH_4^+-N), total oxidized nitrogen (NO_3^-N+ NO_2^-N), and ortho-phosphate (PO_4^{-3}-P) concentrations. Concentrations of major dissolved anions, including SO_4^{-2} and Cl^-, will be determined via ion chromatography. Dissolved organic carbon (DOC) will be quantified using an organic carbon analyzer according to Standard Methods (APHA 2005).

Channel and riparian characteristics will be determined as outlined by Fritz et al. (2006). Points will be taken at each end of the reach (0 m; 60 m), and distance between these points will be used as the valley distance. Sinuosity will be calculated as the quotient of thalweg distance (60 m) to valley distance.

The area of each catchment will be quantified from flow direction analysis of the most recent digital elevation model (DEM) available, ensuring that these models reflect post-mining contours. The downstream end of the reach will be identified in the field and used as the pour point for flow direction analysis. This raster will then be used to clip the most recent land-use/land-cover (LULC) data, and catchment level variables will be quantified for each local catchment. These data will be checked against recent aerial photography to ensure their validity. Land-use types will then be calculated as percent of total watershed area and used as explanatory variables.

Benthic macroinvertebrate assemblage. Benthic macroinvertebrate assemblage structure will be evaluated using samples obtained via a flow-independent protocol outlined by Fritz et al. (2006). Specimens preserved in 95% ethanol will be identified to genus/lowest practicable level and assigned to FFGs using Merritt and Cummins (1996) as well as discriminatory methods based on mouth parts. Several assemblage structure statistics will be calculated, including: shredder density, benthic macroinvertebrate density, and Ephemeroptera-Plecoptera-Trichoptera (EPT) richness. All density measures will be reported as number of individuals per g of litter remaining as AFDM. Non-aquatic taxa and life stages will be eliminated from samples and will not be used in further analyses.

Riparian litter inputs. Energy inputs to stream ecosystems occur through one of three vectors: biologic, meteorologic, or geologic (Fisher and Likens 1973). We are concerned primarily with meteorologic inputs of litterfall. Because extensive sampling of riparian inputs within or upstream of the reach may influence the ambient biotic assemblage, litter inputs will be estimated using 10 direct-fall traps (five on each bank) spaced equidistant across a 30 m sub-reach downstream of leaf bag deployment sites. Inputs from these traps will be assumed to accurately represent meteorologic inputs of litter from the riparian assemblage where litter bags will be deployed. Direct-fall traps are being constructed using 5-gal. buckets lined with mesh screen for drainage. Traps will be placed in September 2010, emptied at least monthly for the first two months and bimonthly through July 2012. Total mass of input, species-specific relative contribution of trees to input, and input rate (a function) will be determined.

Functional Measures

Leaf litter breakdown. A standard substrate is necessary to assess relative stream condition based on leaf litter breakdown (Wieder and Lang 1982). *Quercus alba* leaves will be used as the standard substrate in these streams. Q. *alba* leaves will be collected from a single location using aerial litter traps constructed of plastic netting as described in Crowl et al. (2006) during September and October of 2010. *Q. alba* was chosen because it is common to many of the selected catchments for study, readily available, can be processed without undue concern for loss of coarse fragments (unlike *Tsuga canadensis* for example), and has been used in a previous study for assessment of mining effects on ecosystem function (viz., Fritz et al. 2010). In addition, because *Q. alba* is a "slow processor" (Petersen and Cummins 1974; Webster and Benfield 1986; Benfield 2006), periodic retrieval is logistically feasible in comparison with "fast processors". Leaves will be uniformly mixed, air dried (~20° C), and repeatedly "fluffed" until relatively constant dry mass has been achieved to avoid unrealistic degradation of organic matter (Boulton and Boon 1991). This process could take ca. 5-8 d (Benfield 2006) to ca. 30 d (Fritz et al. 2010).

Eight grams of leaves will be weighed and placed in ~28 cm x 30 cm nylon mesh bags. In accordance with recommendations of Young et al. (2008), both coarse (6 mm) and fine (0.5 mm) meshes will be used. Breakdown in coarse mesh is an index of leaf litter processing due to macroinvertebrates, microbial action, and physical processes, whereas fine mesh excludes any processing due to macroinvertebrates. Thus, assuming weight loss due to physical fragmentation and leaching is minimal, breakdown in fine mesh serves as an indicator of microbial activity in leaf packs. Twelve bags of both coarse and fine mesh (24 total) will be deployed in each 30 m upstream sub-reach in late October of 2010, and will be anchored in glides (transitional zones between riffles and pools) using gutter nails or other means where bedrock prohibits. Because some of our study streams are intermittent, care must be taken to ensure that bags are placed in areas with maximum likelihood of remaining inundated for the entirety of incubation. Multiple authors (Webster and Benfield 1986; Boulton and Boon1991; Young et al. 2008; Fritz et al. 2010) recognize the effect leaf pack placement has on leaf litter decomposition. Although reviews of leaf litter breakdown methods often stress the importance of placement of packs where breakdown in packs would approximate natural conditions, Boulton and Boon (1991), Young et al. (2008), and Fritz et al. (2010) opt for consistent deployment of packs in pools as a safeguard against fluctuation in hydrologic permanence (i.e., drying) of ephemeral and intermittent channels. However, breakdown rates may not be representative of typical processing in pools because they are often relatively hypoxic areas of sediment deposition, and shredders may not be well represented. We will use glides as the standard habitat unit as compromise between representative characterizations of leaf litter breakdown processes and safeguarding against drying of the channel. Leaf bags will be deployed in consistent habitat types across all sites.

Bags will be collected at 0 d, ~30 d, 90 d and ~150 d (Benfield 2006). Including the initial sampling instance at 0 d, bags will be collected in triplicate, taking care not to damage or lose fragments, placed into Ziplocs, and stored on ice for transport back to the lab. Leaves will be removed from the Ziplocs and mesh bags, and rinsed over 250 μm sieves to separate litter from mineral deposits and macroinvertebrates. Leaves from triplicate samples will be placed in separate paper bags to air dry (~20°C) to constant dry mass (DM), and subsequently triplicate samples will be aggregated, milled, and ashed at 550°C in previously ashed and weighed pans. Percent organic matter will be calculated and multiplied by DM to obtain ash free dry mass (AFDM), and percent AFDM remaining will be determined as described by Benfield (2006). The processing coefficient (*k*) will be determined based on a first-order decay model of the form, $m_t = m_0e^{-kt}$, where m_t is the percent AFDM remaining after time, t, m_0 is the initial percent

AFDM (from 0 d bags), and t is time in days or degree-days. These breakdown coefficients will be used to apply the index proposed by Gessner and Chauvet (2002; Table 2).

Table 2. Framework for assessing stream functional integrity from leaf litter breakdown (from Gessner and Chauvet 2002).

Method	Assessment parameter	Criterion	Score
Comparison with reference	Ratio of breakdown coefficients at impacted (k_i) and reference (k_r) site	k_i:k_r = 0.75–1.33	2
		k_i:k_r = 0.5–0.75 or 1.33–2.0	1
		k_i:k_r < 0.5 or >2.0	0
Absolute value	Breakdown coefficient at impacted site (k_i)	k_i = 0.01–0.03/d	2
		k_i = 0.005–0.01/d or 0.03–0.05/d	1
		k_i < 0.005/d or >0.05/d	0
Absolute value of ratio	Ratio of breakdown coefficients in coarse (k_c) and fine (k_f) mesh bags†	k_c:k_f = 1.2–1.5	2
		k_c:k_f = 1.5–2.0 or <1.2	1
		k_c:k_f > 2.0	0

† If sizable numbers of shredders are predicted to occur in the stream.

Benthic microbial biomass accrual rates. The periphyton assemblage is composed of both heterotrophs and autotrophs. Periphytic biomass, as distinct from exclusively algal biomass, may therefore be indexed by ash free dry mass (AFDM) of a sample. Consequently, benthic algal standing crop may be overestimated by AFDM determination. Determination of chl *a*, the sole photopigment common to all algal taxa, provides an alternative to AFDM when quantification of algal standing crop is of interest.

Accrual rates of periphyton on sterile substrate incorporate rates of colonization, growth, as well as loss rates due to scour, sloughing, and grazing. Gross primary production (GPP) is equivalent to the sum of community respiration (CR) and net primary production (NPP). Primary production is the rate at which C is fixed by autotrophs when corrected for loss due to respiration of these primary producers. Biomass accrual rates, measured as changes in chl *a* concentrations over time, should therefore be roughly equivalent to NPP in these streams when effects of scour, sloughing, and herbivory losses are controlled.

Artificial substrates will be used to measure periphyton biomass accrual (AFDM) rates and algal biomass (as chl *a*) accrual rates. Though artificial substrates do not accurately replicate natural conditions in streams, if standardized, reproducible results have been more readily obtained from artificial substrates than from natural substrates (Tuchman and Stevenson 1980). Relatively high levels of reproducibility facilitate comparison among streams differing in substrate composition. Ceramic tiles glued to blocks will be deployed in the upstream sub-reach of each site, arrays will include a total of 20 tiles per reach; ten tiles each for chl *a* and AFDM determination. Arrays will be retrieved following *in situ* incubation for approximately one month during September and October of 2010 and 2011.

In a review of periphyton field methods, Aloi (1990) noted that several incubation periods ranging from 1 d to several years have been used, though exposure periods of two weeks and one month are standard periods of exposure. Trophic status has been used to determine the appropriate incubation period, with more highly oligotrophic (i.e., nutrient-poor environments) environments requiring longer times for incubation (Aloi 1990). Further complicating the choice of incubation period are factors such as grazing, scour, and the purpose of the investigation. The longer the substrate is exposed to the environment, the more susceptible is the periphytic assemblage to grazing and scour; thus, biomass accrual rates as estimates of NPP may become less reliable over time. Because the purpose of this study is a comparative analysis of biomass accrual rates, rather than finer resolution taxonomic comparison, successional development stage (e.g., mature vs. developing) of the assemblage may not be a necessary criterion for selecting the appropriate exposure. The dependent variable (biomass accrual rates) is of relatively coarse resolution, and given that mountain headwater streams are relatively nutrient poor compared to downstream reaches, an incubation period of one month will be used.

Although Kevern et al. (1966) suggest instantaneous growth rate collected at two- to four-day intervals in laboratory streams as a better estimator of primary productivity than mean growth rate, retrieval of periphytometers across such a large spatial extent is not practicable. The alternative of a shorter incubation period defeats the practical nature of this investigation—to evaluate functional parameters that integrate changes over a relatively large temporal scale.

Concentrations of chl *a* on substrata will be determined using either flourometry, spectrophotometry, or high-performance liquid chromatography (HPLC) according to standard methods (APHA 2005). Algal biomass accrual rates will be determined by dividing final chl *a* concentration by incubation time (two weeks to one month). Periphyton biomass accrual rates will be determined by the quotient of AFDM and incubation time. Additionally, the ratio of AFDM to chl *a* will be used to determine the autotrophic index (AI) for each site as a potential indicator of carbon cycling and relative ecological condition of each stream.

Data Analysis.

Given that the majority of restored/reconstructed streams encountered in the central Appalachian coalfields are typically $\leq$ 5-7 years old, multiple regression and multivariate techniques will be used to elucidate factors that may control C function of these streams. Though a more comprehensive age distribution is ideal, sufficient variation is expected to exist among streams to determine factors controlling functional variation among streams. Comparison of restored streams to relatively undisturbed reference conditions will facilitate the evaluation of C dynamics relative to a standard.

With regard to the stream population chosen, we will assemble a set of continuous independent variables (Table 3), which may influence both C function and biotic structure. Because measures of richness, density, evenness, and function are continuous dependent variables, regression and multivariate techniques are well suited to this study. In addition, we will have categorical benthic macroinvertebrate taxonomic data and streams will be divided into a minimum of two *a priori* classes (e.g., UFR and MRR; Table 2). Correlations among continuous variables will be explored to determine relationships between explanatory (independent) and response (dependent) variables. In addition, the means of structural and functional variables will be compared among *a priori* classes using two-way analysis of variance (ANOVA; SAS v.9.1.3, SAS Institute, Cary, North Carolina). Using catchment as the first factor and *a priori* class as the second, results will be used to determine differences among classes of stream given that they occur in different catchments. Bonferroni correction (α/N) will be applied to determine significance for results of multiple pairwise comparisons.

Table 3. Continuous independent and dependent variables to be used in regression and multivariate analysis, as well as *a priori* stream and catchment categories.

Independent Variables (x)	Dependent Variables (y)
Reach Scale Continuous	**Benthic Macroinvertebrate Structure**
Total alkalinity (mg L^{-1} as $CaCO_3$)	Total density (#/g litter remaining)
TDS (mg L^{-1})	Total Taxon richness
TSS (mg L^{-1})	EPT taxon density (#/g litter remaining)
SRP (mg PO_4-P L^{-1})	EPT taxon richness
DIN (mg L^{-1})	Shredder density (#/g litter remaining)
DOC (mg L^{-1})	Shredder richness
Dissolved NH_4-N (mg L^{-1})	Diversity Index Scores
Dissolved NO_2+NO_3-N (mg L^{-1})	
Dissolved Mn (mg L^{-1})	**Riparian Inputs***
Dissolved Fe (mg L^{-1})	Annual litterfall input rate (g AFDM m^{-2} y^{-1})
Dissolved Ca^{2+} (mg L^{-1})	Annual litterblow input rate (g AFDM m-2 d-1)
Dissolved Mg^{2+} (mg L^{-1})	Relative contribution by species (g leaf species / g total)
SO_4^{2-} (mg L^{-1})	
Br^- (mg L^{-1})	**Leaf Litter Breakdown**
Cl^- (mg L^{-1})	$\lvert k \rvert$ (d^{-1} and degree d^{-1}) **
Specific Conductance (µS cm^{-1})	$k_i : k_r$
Dissolved Oxygen (mg L^{-1})	$k_c : k_f$ ***
Temperature (°C)	
pH	**Biomass Accrual**
Canopy cover	Periphytic biomass accrual rate (mg AFDM cm^{-2} d^{-1} and degree d^{-1})
Sinuosity (thalweg dist/channel dist)	Algal biomass accrual rate (mg chl *a* cm^{-2} d^{-1} and degree d^{-1})
Channel slope (%)	Autotrophic index (mg AFDM cm^{-2} / mg chl *a* cm^{-2})
D_{50} (mm)	
Catchment Scale Continuous	
Age since mined (yr)	
Age since restored (yr)	
Age since restored (yr)	
Mining extent (% catchment mined)	
Forest extent (% forested)	
Reach and Catchment Scale Categorical	
UFR	
URI	
MRR	
Restoration Type (e.g., NCD vs. Other)	
Mining type (e.g., VF vs. strip vs. deep)	

*Riparian inputs will also serve as potential explanatory variables with respect to leaf litter breakdown and assemblage structure

** Will be calculated for all sites, but will only applyGessner and Chauvet (2002) index to those which are not forested reference (UFR).

*** Only for standardized (*Q. alba*) leaf bags

Conclusions

Based on observations from scouting study sites, and communications with several contacts, it is clear that there are a wide range of methods to construct and restore streams for mitigation purposes. In this context, development of this study has provided a clear framework to achieve the objectives outlined. Moreover, this research will benefit the scientific community, regulatory agencies, monitoring authorities, and industry by: (1) increasing the body of knowledge associated with functional assessment of stream ecosystems following restoration practices in areas subjected to mining activities, (2) investigating methods that may be applied for accurate assessment of the functional status of these streams, (3) assessing effectiveness of stream restoration efforts and the relationships between structural and functional integrity within these stream ecosystems, and (4) attempting to guide future functional assessment through dissemination of results via presentations at professional meetings and publication in peer-reviewed journals, as well as cooperation with industry, regulatory agencies, and consulting firms.

Future Directions

The C functions measured in this study are only components of the entire ecosystem. Future study may involve augmenting or changing the list of ecosystem processes and/or structures measured to obtain a more holistic view of these streams. Furthermore, long-term study of these and other processes may be warranted to track ecosystem recovery over time and to gain insight to the factors affecting the development of these systems. This approach has potential for determination of restoration/reconstruction “success trajectories” based on structural and functional characteristics of the system.

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POWELL RIVER PROJECT

Progress report

Herbaceous Crops for a Biofuels/Bioproducts Industry on Reclaimed Mine Lands

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Summary

Biofuel species comparisons were begun at the Powell River Project in 2007 to determine yield capacity of several feedstock species with potential suitability for revegetating mined land. Feedstock treatments included panicgrass, switchgrass, a 1:1 seed mix of panic- and switchgrasses, and two species established from vegetative propagules: hardy sugarcane and miscanthus. Plants were established on mined lands near the Powell River Project Research and Education Center on 30 May 2007. The original miscanthus species was mis-identified and thus removed and replanted in 2008. Establishment conditions were difficult due to drought, but about 70% of plants set out the previous season had new growth as of May 2009. Dead plants were replaced in May 2009. Initial survival and harvest data from the 2008 growing season suggested miscanthus might be well-suited to production on mine lands. Hardy sugarcane production has been marginal, and the plant has high losses on the worst soils at the site. Yield data for the 2009 growing season are not reported. Miscanthus yields in 2009 were uniform across plot replicates, but less than anticipated. Stand loss from mid-November to late-March averaged about 9% for miscanthus, compared with 46% for all seeded grass species. Both switchgrass first harvest yields (1.75 ton/acre) and over-winter losses (55%) were greatest among seeded species. Native grasses such as switchgrass currently appear to be front runners for bioenergy crops on reclaimed mine sites.

I. Introduction:

Renewable energy sources and bio-based chemicals are of increased interest for national, environmental, and economic security and rural development. No other new agricultural enterprise has the potential for such large impact as the biorenewables industry, both in terms of economics or land-use change. The scale of the endeavor and the potential competition for crop land heightens the need to use marginal lands to avoid competition between food and fuel production. Finding species that are productive under mined land conditions may represent a win-win situation in meeting the nation's energy goals and utilizing the large marginal sites.

Yield per land area will be one of the most important determinants for economic viability of a biomass-to-biorenewables industry. Because raw biomass has low value (in dollars per ton)as a commodity, output per acre must be sufficient to warrant investment in growing the crop. Given the extensive nature of sometimes difficult terrain on mined lands, these sites must also be productive with minimal inputs.

II. Objectives:

1. Evaluate and compare stand establishment of potential biofuel/bioproduct crops (switchgrass, coastal panicgrass, and a mix of these two native grasses, along with two non-natives, miscanthus, and hardy sugarcane) on reclaimed mine lands in Southwest Virginia.
2. Determine yield as a function of harvest time.
3. Examine feedstock quality (cellulose, hemicellulose, lignin, nitrogen, and ash) of these potential biomass crops.
4. Determine the carbon sequestration potential of these biomass crops.

III. Methods and Procedures:

Plant species: Switchgrass, coastal panicgrass, and a 1:1 mixture of these species were seeded into plots with a plot seeder on 30 May 2007. At the same time, 100 plants/plot were established for both miscanthus and hardy sugarcane. Subsequent research determined that the miscanthus species planted in 2007 was not the species intended, and these plants were killed out and replaced in summer of 2008. In addition, switch- and panic grasses were overseeded in one replicate in 2008 due to stand loss from frost heaving.

Plants were not fertilized with N in 2007 or 2008 to help reduce weed competition. Fertilizer (50 lb N/ac) was applied in 2009.

Measurements: Stand counts and plant growth measurements such as height, crown width, and tiller number were in October 2007 to determine initial production. Biomass samples were collected in January 2008, and plots also were evaluated for frost heaving. In March 2008, plots were evaluated for winter kill.

Full-scale biomass harvests were conducted in Fall-Winter ‘08–’09 (three harvests) and ’09-’10 (two harvests).

IV. Brief progress report:

Hardy sugarcane, which was successfully established under drought conditions in 2007 has nearly disappeared from some plots due to frost heaving or winterkill or both (greater than 50% in two of three reps). The plant notably stays green at the base for much of the winter, likely increasing its sensitivity to cold temperatures. Sugarcane yields were marginal at best and considered nil in 2008. While we continue to measure those plants which survive, the limited plant material – coupled with the large death losses – leads us to abandon this species for use in mineland energy cropping. Production from 2008 was minimal and data are not reported.

Miscanthus yields following the establishment year (2008-2009) were about 0.5 tons/acre at the first harvest, with subsequent yields of about 0.24 tons/acre during winter and early spring. Yields declined largely due to leaf drop over the winter. This characteristic may improve feedstock quality by keeping minerals and silica (largely found in leaf tissues) out of the feedstock supply. However, 50% yield reductions probably would not offset the anticipated gains in feedstock quality. Yields following the 2009 growing season were lower than for the preceding year, and although the plants are not being lost as with sugarcane, it remains to be seen whether miscanthus will be viable on mined lands.

Panic and switchgrasses and the mixture of the two had great productivity gains from the 2008 to 2009 growing seasons. First-harvest yield for the three species treatments was similar (average = 0.55 tons/acre) following the 2008 growing season but declined more in switchgrass and mixed plots (average = 0.33 tons/acre) than for panicgrass (0.52 tons/acre).

Much larger yields were observed for these plots following 2009 (1.5 tons/acre), but the pattern of larger declines for plots with switchgrass was again evident. A mid-season harvest was not possible this year due to snow cover or wet soils for much of the winter. Switchgrass yields were greatest among species treatments at harvest 1 (1.75 tons/acre) but declined the most (55%) to 0.79 tons/acre (~55%) by early spring. Panicgrass was less productive (1.26 tons/acre) compared with switchgrass, but losses over the winter were less (~38%) resulting in the same yield by early spring (0.79 tons/acre). The mixture was intermediate in terms of first yield (1.37) and over-winter losses (47%).

We anticipate collecting another season of production data and beginning feedstock analyses this fall. Soil data are to be collected after this season to determine feedstock effects on C sequestration.

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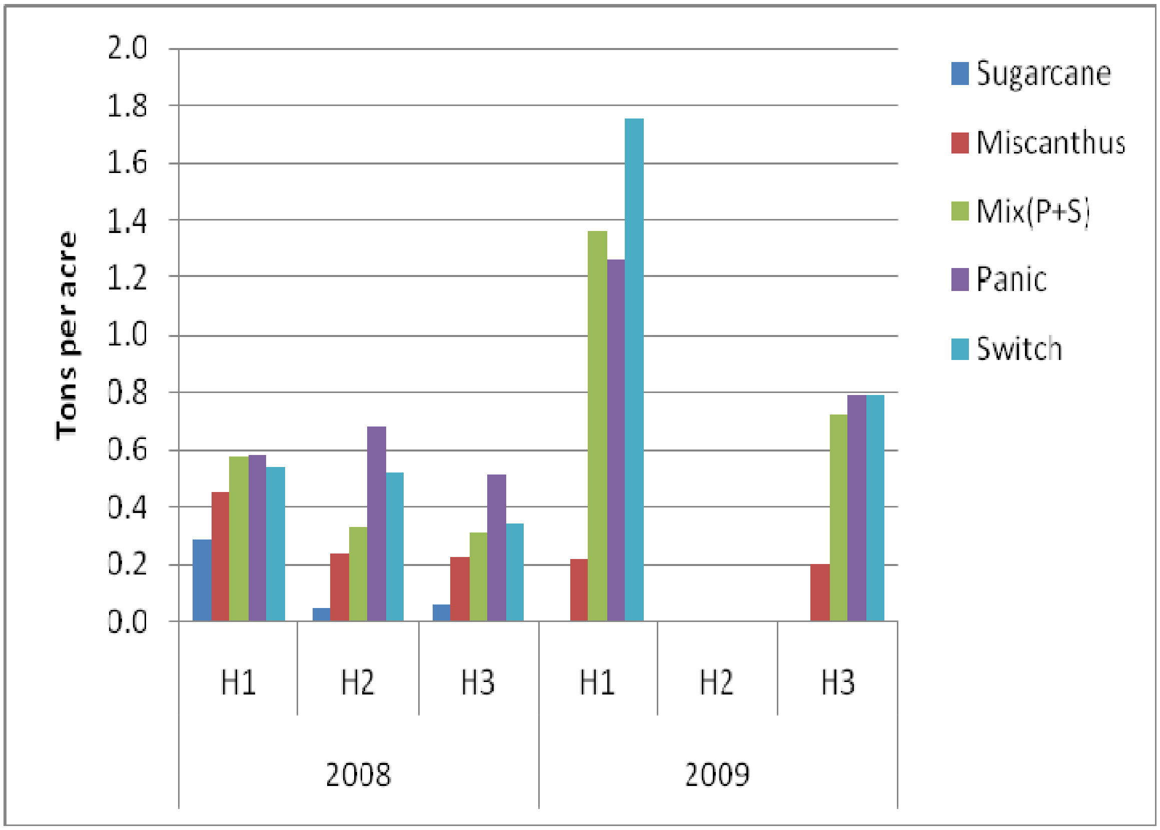

Figure 1. Biomass yields for each of three harvests (H1. H2, and H3) in 2008, and for two 2009 harvests (H1 and H3). Yields are on a dry-matter basis.