CHARACTERIZATION OF NATURAL WETLAND SOILS RECEIVING ACID MINE DRAINAGE[1]

Kyle M. Stephens, John C Sencindiver, and Jeffery G. Skousen[2]

Abstract. A proposed section of Appalachian Corridor H will pass through an area of the Beaver Creek watershed in Tucker County, WV previously mined for the acid-producing Upper Freeport coal. Presently, partially-reclaimed spoils from past mining activities are generating acid mine drainage. Wetlands adjacent to the spoils are supporting vegetative communities and they appear to be naturally treating the drainage. We characterized soils of the acid mine drainage-impacted wetlands and non-impacted wetlands to establish baseline soil status and to compare the acid mine drainage-impacted soils and non-impacted soils. We analyzed soil samples for pH, sulfate, acid volatile sulfides, chromium reducible sulfides, organic sulfur, total sulfur, total carbon, and total iron. Platinum redox electrodes were placed in some of the wetland soils to assess near surface redox chemistry. Wetlands receiving acid mine drainage generally contained higher levels of total sulfur, and the sulfur in these wetlands was more likely to be in the form of acid volatile sulfides, sulfate, and/or chromium reducible sulfides. Chromium reducible sulfides ranged from 0.0 to 103.4 *u*mol g^{-1} and acid volatile sulfides ranged from 0.0 to 32.3 *u*mol g^{-1} in all wetlands. The carbon content in wetlands receiving acid mine drainage was generally lower than the non-impacted wetlands and the iron content was generally higher in the acid mine drainage-impacted wetlands. Redox values indicated that conditions are present in impacted wetlands for iron and sulfur reduction to occur.

Additional Key Words: Acid volatile sulfides, Chromium reducible sulfides, Beaver Creek Watershed, Redox potential

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[2] K.M. Stephens is graduate research assistant and J.C. Sencindiver and J.G. Skousen are Professors in the Division of Plant and Soil Sciences, West Virginia University, P.O. Box 6108, Morgantown, WV 26505-6108.

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Introduction

Acid mine drainage (AMD) is an acidic, sulfur-rich runoff that forms upon exposure of pyrite (FeS_2) or other sulfidic materials to oxygen (O_2) and water (H_2O). Though the chemistry is variable, AMD typically has high iron (Fe) concentrations in the range of 39 to 7300 mg/L and a pH range of 2 to 4.5 (Appalachian regional Commission, 1969). Furthermore, elevated sulfate (SO_4^{2-}) concentrations are often associated with AMD (Skousen et al., 2000).

Wetlands receiving AMD often will be negatively impacted. Studies have shown that Fe and sulfur (S) concentrations will increase in wetlands receiving AMD (Wieder and Lang 1986: Tarutis and Unz, 1996). Despite these impacts, wetlands have been shown to improve the quality of AMD passing through them (Wieder, 1993; Stark et al., 1996; Tarutis and Unz, 1996). When AMD enters a wetland, Fe is retained, mainly in the organically bound reduced form or the oxidized hydroxide form (Wieder and Lang, 1986), and SO_4^{2-} is reduced to the form of organic sulfur, sulfides, or to the elemental state (Laudon, 1989). Upon reduction of Fe and S, alkalinity may be generated, thereby raising the wetland pH (Vile and Wieder, 1992). For reduction of Fe and S to occur, certain conditions must be present: enough organic matter must be readily available as an energy source for microbial reduction and the wetland soil redox potential (Eh) must be low enough.

Sulfur in AMD is generally in the form of SO_4^{2-} when it enters a wetland. If the proper conditions are present, sulfur will be microbially reduced to iron monosulfides, (FeS), iron disulfides (FeS_2), elemental sulfur (S°), or organic sulfur. The presence of FeS is indicative of recent S deposition, and the presence of FeS_2 and S° is a sign of longer-term deposition. Studies have shown that for freshwater wetlands, most sulfur exists in the organic form, but in AMD impacted wetlands, more sulfur will take on the form of FeS or FeS_2 (Wieder and Lang, 1986).

Redox potential is a quantitative measurement of a soil's ability to oxidize or reduce certain elements. It can be measured in the field using platinum electrodes, thereby allowing understanding of the valance state of certain elements. Iron will normally be reduced at Eh values between 100 to –100 mV, and S will normally be reduced at Eh values between –100 to –200 mV (Mitsch and Gosselink, 2000). The variation is dependent on soil pH with reduction occurring at higher redox potentials when the soil pH is lower.

Beaver Creek Watershed is an area of Tucker County, WV that has been mined extensively for the acid-producing Upper Freeport Coal. As a result, many partially reclaimed mine spoils are situated throughout the watershed. Some of these mine spoils release AMD and sediments into nearby wetlands. Wetlands receiving AMD appear to be treating the drainage and improving water quality. The West Virginia Division of Highways (WVDOH) is planning to construct a portion of Appalachian Corridor H through this sensitive watershed, but intends to minimize any impacts to existing wetlands during and following highway construction. Furthermore, the WVDOH plans to use the existing wetlands as models for construction of AMD treatment wetlands following road construction.

The first objective of this study was to assist the WVDOH in evaluating AMD-impacted and non-impacted wetland soils along the proposed route of Corridor H so that a baseline wetland soil status could be established. The second objective was to evaluate the conditions in the wetland soils and compare the AMD-impacted and non-impacted wetlands.

Methods

Site Descriptions

All wetlands were located near the proposed route of Corridor H, which will replace the existing Rt. 93 (Fig. 1). Four wetlands were sampled in this study and are identified as: Narrow Wetland, Iron Pond, Railroad Grade, and Elder Swamp (Fig. 1). Narrow Wetland, Iron Pond, and Railroad Grade are situated adjacent to mine spoils generated during the mid 1960's. These wetlands are also adjacent to Rt. 93, which was constructed about 30 years ago.

Narrow Wetland was a relatively flat wetland located on the southeastern side of Rt. 93 (Fig. 1). The soils appear to have been impacted by adjacent, partially- reclaimed Upper Freeport coal spoils, located on the wetland's southwestern side. Cattails (*Tyhia latifolia*), sedges (*Carex* spp.), sphagnum (*Sphagnum* spp.) and hair-cap mosses (*Polytrichum* spp.) were the prevalent vegetation. Textures of the mineral soil horizons below the organic layer were silt loam or silty clay loam.

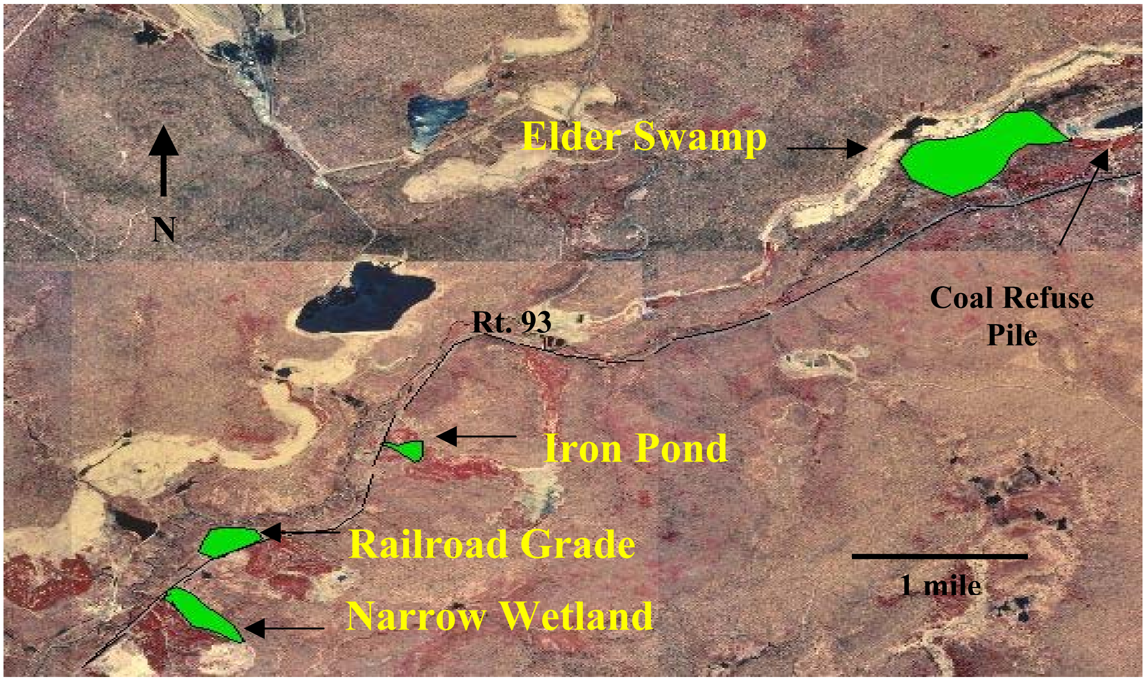

Figure 1. Aerial photograph of wetland sites in the Beaver Creek watershed, WV. Wetlands are green areas, mine spoils are dark red areas.

The organic layer for all wetland soils was divided into Oi, least decomposed, Oe moderately decomposed, and Oa, most decomposed. The average thickness of these layers and the average total thickness of the organic layer for each wetland are presented in Table 1. Narrow Wetland contained a 16-cm thick organic layer (Table 1).

Iron Pond is the youngest, smallest, and by far the most AMD-impacted wetland in the study. Steep, partially-reclaimed Upper Freeport coal spoils surrounded this wetland. Iron Pond was receiving AMD from multiple seeps along the base of the surrounding spoils. A small stream dissected this wetland before it spilled into an orange pond that was impounded by Rt. 93 and surrounding mine soils. The prevalent vegetation in Iron Pond was cattails, mosses, and sedges. The organic layer had an average thickness of 19 cm (Table 1). Textures of the mineral layers were silt loam, loam, or clay loam.

Railroad Grade was located on the northwestern side of Rt. 93 and was less impacted by AMD than Iron Pond and Narrow Wetland (Fig. 1). Sediments and AMD from the Upper Freeport coal spoils located near the eastern end of Railroad Grade, directly across Rt. 93, appear to have impacted two of the seven soil sampling locations. The rest of the soils do not appear to have been affected. Mosses dominated this wetland and sparse populations of sedges and cattails were scattered throughout. The organic layer was 33 cm thick (Table 1). The mineral layers consisted of silty clay loam material near the surface and silt loam material in the subsurface.

Table 1. Thickness of individual organic horizons (Oi, Oe, and Oa), and total thickness of the organic layer for wetland soils of the Beaver Creek Watershed. Values expressed as a mean of all sampling locations in each wetland, with n representing the number of soil sampling locations the horizon contained.

Wetland		Oi			Oe			Oa			Organic Layer	
	n	Mean	Range	n	Mean	Range	n	Mean	Range	n	Mean	Range
		cm			cm			cm			cm	
Narrow Wetland	5	12	3--42	3	10	6--17	0	--	--	6	16	3—42
Iron Pond	5	18	7--23	1	2	--	0	--	--	5	19	9—23
Railroad Grade	7	16	7--33	7	14	7--48	7	14	5--24	7	33	17--54
Alder Thicket	0	--	--	3	4	3--7	2	16	16--16	3	20	7—34
Upland Soils	0	--	--	2	5	4--7	3	5	2--9	4	7	4—9
Bog-Marsh	4	13	4--23	6	18	7--38	6	70	57--99+	6	98	73—120+

Elder Swamp, the largest wetland in the study area, has been classified as an area of Exceptional Resource Value and as a Wetland Special Area (West Virginia University 1995). This wetland was to act as the unimpacted control, but it appeared that some areas had been impacted by drainage from a nearby coal refuse pile. Elder Swamp was situated on the northwestern side of Rt. 93, east of Iron Pond (Fig. 1). Four wetland areas were sampled in Elder Swamp: an alder thicket, a forested wetland, a scrub-shrub wetland, and a bog-marsh. The organic layer in the alder thicket was 20-cm thick, (Table 1) and the texture of the mineral layers was silty clay loam or silt loam. The forested wetland was located in a thin stand of white pines (*Pinus strobus*), slightly upslope from the alder thicket and the bog-marsh. The scrub-shrub wetland was located next to the forested wetland, and was covered with mosses, blueberry (*Vaccinium* spp), and St Johns wort (*Hypericum cyathifloru*). Soils in these latter two wetland types were similar in nature so they were grouped together for the purposes of this study and will be referred to as upland soils. The average thickness of the organic layer in the upland soils was 7 cm (Table 1), and the mineral soil layers had silt loam or loam textures. The bog-marsh area was the dominant ecosystem, and was the portion of Elder Swamp that appears to be receiving drainage from a nearby coal refuse pile. In the bog, sphagnum and hair-cap mosses cover the ground with a few sedges present. The bog bordered the marsh, which made up the central portion of Elder Swamp. The marsh was dominated by cattails growing on hummocks of peat. A braided stream that fed nearby beaver ponds flowed through the marsh area. The bog-marsh had a 98-cm thick organic layer (Table 1), so these soils would be classified as Histosols. The mineral soil layers had silty clay loam, loam, or sandy loam textures.

Sampling Methods

Nine transects were established in the four wetlands of the Beaver Creek Watershed in the summer of 2001. On each transect, soil description and sampling locations were identified every 50 or 100 m, depending on the size of the wetland. In total, 25 sampling locations were included in the study and each soil was described according to standard soil survey procedures (Soil Survey Division Staff, 1993). The number of soil samples collected in each wetland varied depending on the wetland size. A Dutch auger and peat sampler were used to expose soils for description purposes and to remove soil samples. Sampling was based on soil profile descriptions, with organic horizons sampled by type (i.e., Oi sampled together, Oe sampled together, Oa sampled together) and subsurface mineral horizons sampled by depth: 0 to 10 cm, 10 to 30 cm, 30 to 60 cm, and 60 to 100 cm. The mineral layers are represented by M1, M2, M3, and M4, with M1 being the uppermost layer and M4 being the deepest layer.

Soils were sampled at three locations in Narrow Wetland, five locations in Iron Pond and seven locations in Railroad Grade. In Elder Swamp two sampling sites were located in the alder thicket, four in the upland soils, and four in the bog-marsh. The differences in the number of sampling locations reflected the differences in the size of the wetlands.

Laboratory Analyses

Soil samples were collected and placed in a nitrogen environment at 4°C, and transported to West Virginia University where they were frozen. This was done to maintain sulfur pools in their current field conditions. Subsamples of the frozen soils were removed in the fall of 2001, air-dried and passed through a 2-mm sieve for analyses. Soil samples were analyzed for total carbon and total sulfur on a LECO CNS 2000 analyzer. Soil pH was measured in a 1:1 soil:water suspension using a silver/silver chloride double junction electrode. Many of the soil samples with high carbon values required additional water to be added to the pH soil:water suspensions. Additionally, cation exchange capacity, total nitrogen, relative particle size, exchangeable bases, extractable aluminum, and electrical conductivity were analyzed as part of a larger study, but are not reported in this paper.

Sulfur was fractionated into SO_4-S, FeS-S, FeS_2, and organic sulfur for all frozen soil samples. We used methods reported by Urlich et al. (1997) and Kennedy et al. (1999) as a basis to develop our procedure. Our procedure is different from the methods Urlich et al (1997) and Kennedy et al. (1999) described, so it has been reported in detail below. To obtain FeS and FeS_2 the frozen soils were kept at 4°C until thawed and then three subsamples were removed. The first sample was used to obtain moisture content. The second was extracted with sodium phosphate to obtain SO_4-S, which was measured spectrophotomectircally on a Lachate flow injection analyzer. The last subsample was placed in a 60-ml glass vial to obtain FeS-S and FeS_2-S. Next, 2.5 ml of deoxygenated 1 M sodium hydroxide (NaOH) was injected into a 6-ml plastic vial that was fitted with a rubber o-ring. The vile was then placed in the 60-ml glass vial that contained the soil sample. A lid with a Teflon coated septa was placed on the vial, and the vial was deoxygenated for 60 seconds with 25 psi of nitrogen gas (N_2). Following this, 3-5 ml of deoxygenated 6 N hydrochloric acid (HCl) was injected into the vial and the vial was placed on an oscillating shaker at 130 rpm for 48 hours. All sulfide released during this time was trapped in the NaOH solution as sodium sulfide (Na_2S), and was considered to be in the form of FeS. This was not confirmed, so for this paper all sulfide detected in this step will be operationally defined as acid volatile sulfides (AVS).

The vial was opened after 48 hours under N_2 gas flow, never allowing the soil to be exposed to atmospheric conditions, and the NaOH trap was removed. The glass vial was centrifuged, and the supernatant was collected for total iron analysis. A fresh NaOH trap was placed in the vial, and the vial was degassed for 60 seconds at 25 psi with N_2 gas. This time 2-3 ml of deoxygenated concentrated HCl was injected into the vial and 2-3 ml of reduced chromium (Cr^{2+}) in the form of 1 M chromic chloride was injected into the vial. The vial was again placed on the oscillating shaker for 48 hours at 130 rpm. After 48 hours of agitation the traps were removed and the vial was centrifuged. Sulfide collected in this trap was presumably FeS_2 but this was not confirmed, so these sulfides will be referred to as chromium reducible sulfides (CRS). The supernatant was collected and mixed with the supernatant from the first step, so that total iron could be analyzed via the inductively coupled plasma spectrometer. The NaOH traps were brought to a known volume, and were analyzed spectrophotometrically for Na_2S on a Lachate flow injection analyzer.

Lastly, organic sulfur was obtained by subtracting the sum of SO_4-S FeS-S, and FeS_2-S from the total sulfur value. Any elemental sulfur present in these samples would therefore be included in the organic sulfur form.

Using a crushed glass matrix of FeS and FeS_2 standards, Kennedy et al. (1999) reported a 99.6 % recovery rate for FeS and a 105.4 % recovery of FeS_2. Urlich et al. (1997) reported recovery rates greater then 90 % for FeS and FeS_2 crystalline standards. We observed recovery rates over 90 % for crystalline FeS and FeS_2 standards that were run through our procedure.

Redox Measurements

In the fall of 2001, platinum-tipped redox electrodes were constructed according to instructions provided by Faulkner et al. (1989). In December 2001, these electrodes were placed in the ground and the first measurements were taken 24 hours after initial placement. Redox measurements were made in the field for one year using a portable voltmeter and a saturated calomel reference electrode. Meter readings were adjusted by adding +200 mV so redox potential was based on the standard hydrogen reference electrode instead of the saturated calomel electrode. At each redox sampling location, six probes were permanently placed at a 20-cm depth and six at a 10-cm depth. The average of the six electrodes was used as a single redox value. Measurements were taken at three locations in each of the following three wetlands: Railroad Grade, Iron Pond, and Elder Swamp. In Railroad Grade, one location (stop 1) was AMD-impacted and the other two locations were unimpacted. In Elder Swamp, measurements were taken from soils in the bog, marsh and scrub-shrub wetlands. At Iron Pond, measurements were taken from three locations representing three different vegetative communities. All sites in Iron Pond were considered AMD-impacted.

Redox probes at the 10-cm depth were placed in an A horizon in the scrub-shrub site and in organic horizons at all other sites. At the 20-cm depth, the probes were placed in a B horizon at the scrub-shrub site and in A horizons at Iron Pond stop 5 and Railroad Grade Stop 1. At all other sites, the 20-cm probes were placed in organic horizons.

Soil pH also was measured at both 10 and 20 cm when each of the monthly redox readings were taken. To simplify interpretation of Eh readings and comparisons among wetlands, redox potentials were adjusted to represent Eh at pH 7. This was done by subtracting 59 mV for each unit of decrease below pH 7 (Karathanasis et al., 2003).

Results and Discussion

Soil pH, Total Carbon, and Total Iron

The Oi layer in Iron Pond had the lowest average pH (4.2) of any Oi layer in the study (Table 2). Five sites were sampled in Iron Pond and all five contained Oi layers. The pH of the Oi layer at each of the individual sampling locations in Iron Pond showed the following decrease in pH when moving from the head (relatively unimpacted section) to the terminus of this wetland: 6.0, 5.0, 3.4, 3.6, 3.2. This indicates that Iron Pond is retaining hydrogen proton acidity generated from the precipitation of iron hydroxides as the AMD flows through the wetland.

As indicated by the ranges, the pH of the organic layers in Railroad Grade was variable (Table 2). However, when looking at the pH of the individual sampling locations in Railroad Grade, we observed that the lower pH values occurred in the organic layers of the non-impacted portion, and the higher pH values occurred in the organic layers of the AMD-impacted portion. As compared to Iron Pond, less iron hydroxides were observed in the AMD-impacted portion of Railroad Grade; therefore, hydrogen proton generation via iron hydroxide precipitation is not as influential on soil pH in the AMD-impacted portion of Railroad Grade. Furthermore, the generation of alkalinity from reduction of iron and sulfur in the AMD-impacted area may explain the higher pH values observed in the AMD-impacted portion of Railroad Grade (Vile and Wieder, 1992).

In the bog-marsh wetland, the average pH of the Oi layer was 4.5, with a range of 3.8-4.9 (Table 2). The average pH of the Oe layer was 4.8, with a range of 3.9 to 5.7, and the average pH of the Oa layer was 4.7, with a range of 3.8 to 5.9 (Table 2). The wide range in pH of the organic layers in the bog-marsh is related to differences in hydrology. The marsh is located in

Table 2. Soil pH, total carbon, and total iron, for wetland soils of the Beaver Creek Watershed. Organic horizons sampled by type and mineral horizons sampled by depth.

		pH			Total Carbon			Total Iron		
					%			*u*mol g^{-1} dry mass		
					AMD-Impacted Wetlands					
	n*	Average	Range	Stdev	Average	Range	Stdev	Average	Range	Stdev
Narrow Wetland										
Oi	3	5.1	4.1--5.9	0.9	24.1	16.5--28.9	6.7	1504	313--2980	1356
Oe	1	5.6	--	--	28.6	--	--	775	--	--
M1	3	4.9	4.5--5.6	0.6	6.7	5.1--9.0	2.1	199	109--217	83
M2	3	5.0	4.5--5.7	0.6	6.3	5.1--8.0	1.5	105	46--215	95
M3	3	5.2	4.6--6.1	0.7	6.5	5.5--8.2	1.5	66	51--84	17
Iron Pond										
Oi	5	4.2	3.2--6.0	1.2	16.7	9.6--26.0	6.8	1588	644--3709	1269
Oe	1	6.0	--	--	13.2	--	--	1079	--	--
M1	5	5.0	4.7--5.8	0.4	5.1	2.1--8.7	2.9	763	315--1298	374
M2	5	4.7	4.5--5.1	0.2	2.3	0.8--4.2	1.4	480	334--694	132
M3	5	4.5	4.0--4.9	0.4	1.6	0.7--2.4	0.7	428	397--462	24
M4	3	4.4	4.2--4.6	0.2	1.5	1.4--1.8	0.2	533	499--566	33
Railroad Grade										
Oi	7	4.6	3.8--5.9	0.8	33.6	16.8--41.5	9.3	521	40--1350	578
Oe	7	4.5	3.8--5.5	0.5	30.2	16.6--41.2	11.4	467	101--1260	428
Oa	5	4.3	3.9--4.5	0.3	21.1	14.8--31.6	6.8	180	63--380	127
M1	7	4.2	3.9--4.7	0.2	6.6	4.3--9.6	2.2	360	77--1119	354
M2	7	4.2	3.9--4.4	0.2	7.4	1.6--9.8	2.8	265	125--533	147
M3	4	4.4	4.3--4.9	0.3	10.3	6.9--11.0	10.4	472	58--257	242
M4	2	4.3	--	0.0	9.2	2.0--7.5	2.5	543	428--973	608
					Non-Impacted Wetlands					

Table 2 continued

ES: Alder Thicket										
Oe	2	5.4	4.8--5.9	0.8	36.6	32.1--41.1	6.4	685	522-848	230
Oa	2	5.4	5.0--5.8	0.6	27.3	26.0--28.6	1.8	188	71--305	166
M1	2	4.8	4.7--4.8	0.1	9.3	9.2--9.5	0.3	33	31--34	3
M2	2	4.7	--	0.0	9.2	8.1--10.4	1.6	53	34--72	27
M3	2	4.8	4.6--5.0	0.2	7.9	4.6--11.3	4.7	141	131--151	14
ES: Upland Soils										
Oe	2	4.1	4.0--4.3	0.2	37.1	33.8--40.5	4.7	139	109--170	43
Oa	3	3.8	3.6--4.0	0.2	27.1	23.2--31.3	4.1	126	86--185	52
M1	4	3.8	3.7--3.8	0.0	6.4	5.6--8.1	1.2	165	68--253	78
M2	4	3.9	3.9--4.0	0.1	2.5	1.6--3.6	0.8	220	69--415	144
M3	4	4.2	4.1--4.3	0.1	1.4	0.7--1.8	0.8	598	416--1082	323
M4	4	4.6	4.4--4.8	0.2	2.1	0.7--4.6	1.8	480	354--549	142
ES: Bog-Marsh										
Oi	3	4.5	3.9--4.9	0.6	36.8	32.8--42.6	5.1	440	95--855	385
Oe	4	4.8	3.9--5.7	1.0	34.1	18.7--42.8	11.5	337	63--630	303
Oa	4	4.7	3.8--5.9	1.0	36.2	24.9--47.8	10.3	74	48--96	20
M1	2	4.3	3.9--4.7	0.5	4.9	2.2--7.9	4.1	71	29--113	59
M2	1	4.8	--	--	4.8	--	--	45	--	--
M3	1	4.8	--	--	4.9	--	--	58	--	--

* n - is the numebr of sampling locations that contained that layer type

the center of this wetland and the bog is positioned out on the edge of this wetland. When looking at the pH of the individual sampling locations, we observed lower pH values in the bog section and higher pH values in the marsh section. By definition, a bog has no significant surface inflows and outflows (Mitsch and Gosselink, 2000). This means that basic cations are not being replenished and the soil pH is lowered. A marsh does have significant inflows and outflows, allowing for replenishment of pH elevating cations.

The average pH of the organic layers (Oe=4.1 and Oa=3.8) for the upland soils was the lowest in the study (Table 2). These low average pH values resulted from the dry conditions present in these soils, which allows for rapid decomposition of the organic matter. In the alder thicket the average pH of the organic layers was 5.4 for both the Oe and Oa layers (Table 2). The alder thicket soil is much wetter then the upland soils, and as a result, decomposition rates are reduced and less acidity is generated.

In general, the average total carbon in the organic layers was lower in the impacted wetlands than in the non-impacted wetlands (Table 2). The lowest average value observed in any Oi layer (16.7 %) was in Iron Pond, and the range for this layer was 9.6 to 26.0 %. In Narrow Wetland, the average total carbon for the Oi layer was 24.1 %, with a range of 16.5 to 28.9 %. The organic layers in Railroad Grade had higher average total carbon values than what we observed in the Oi layers of Iron Pond and Narrow Wetland (Oi=33.6 %, Oe=30.2 %, Oa=21.6), but the overall range for the organic layers was 14.8 to 41.5 %, with the low end values observed in the AMD-impacted portion of this wetland. The organic layers in the non-impacted areas of Railroad Grade had total carbon values similar to those observed in the organic layers in the Elder Swamp wetlands (Table 2). Wieder and Lang (1986) reported much lower organic matter content in the upper 20 cm of Tub Run Bog, a wetland in West Virginia also impacted by AMD from Upper Freeport Coal spoils, as compared to the non-impacted wetlands they included in their study. They attributed this to the transport of inorganic soil particles from the adjacent mine spoils, which would dilute the soil organic matter content. We believe that this process is lowering the carbon content in Iron Pond, Narrow Wetland, and the impacted portion of Railroad Grade. Furthermore, in Iron Pond and Narrow Wetland, the abundance of iron in the surface layers is also reducing the total carbon content (Table 2).

Average total carbon in the mineral layers was lowest in the upland soils and Iron Pond (Table 2). In the remaining wetlands, average total carbon in any mineral layer was never lower than 4.8 %, indicating carbon is accumulating in the mineral layers of these wetlands. The lower carbon content of the upland soils mineral layers is related to the drier conditions, which favor carbon decomposition, and reduce carbon accumulation in soils. In Iron Pond though, the soils are saturated most of the year, but the carbon content in the mineral layers is still relatively low. The lower carbon content is due to the young age of this wetland soil. This wetland has developed since the construction of Rt 93, roughly 50 years ago. There has not been enough time for carbon to be incorporated into the mineral layers of this wetland.

Average total Fe was highest in the Oi layer of Iron Pond (1504 *u*mol g^{-1}) and the Oi and Oe layers in Narrow Wetland (1588 and 1079 *u*mol g^{-1}, respectively) (Table 2). The standard deviation in these layers indicates variability throughout these two wetlands. In general, individual sampling locations in these two wetlands that appeared to be receiving more AMD had higher Fe concentrations. The values for the Oi layer in Iron Pond and Narrow Wetland were about 1.5 times greater than that observed by Wieder and Lang (1986) in the upper 20 cm of Tub Run Bog, and about three times greater than what we observed in the Oi layer of the bog-marsh wetland (Table 2). Wieder and Lang (1986) fractionated Fe in Tub Run Bog and found that 49 % was in the organically bound form, and 50 % was in the oxide form. Iron pools in Narrow Wetland are more likely to be similar to those reported by Wieder and Lang (1986). In Iron Pond, an abundance of Fe oxides was visible at 4 of the 5 sampling locations, indicating that much of the total iron may be in the oxide form. A separate study by Wieder et al. (1990) showed that 73 to 86 % of the iron retained in a mesocosm wetland designed to treat synthetic AMD over 16 weeks was in the oxide form. They attributed this to rapid saturation of the organically bound exchange sites, leaving oxide formation as the primary means of iron retention.

Average total Fe in the mineral layers of the AMD-impacted wetlands ranged from 66 *u*mol g^{-1} in the M4 layer of Narrow Wetland to 763 *u*mol g^{-1} in the M1 layer of Iron Pond (Table 2). The range for the mineral layers in Elder Swamp wetlands was 33 *u*mol g^{-1}in the M1 layer of the alder thicket to 598 *u*mol g^{-1} in the M3 layer of the upland soils. Yet, in general, mineral layers in the AMD-impacted wetlands had higher Fe content than the non-impacted Elder Swamp wetlands, indicating some Fe from the AMD was accumulating in these subsurface layers (Table 2). Of all the wetlands, the bog-marsh mineral layers had the lowest average Fe content, which explains the lack of AVS and CRS in the mineral layers (Table 3). Iron must be present to form these inorganic sulfides.

Redox Potential

Redox Potential was measured at depths of 10 and 20 cm. This approach was used to observe near surface redox characteristics in some of our wetlands and to relate the Eh to Fe and SO_4^{2-} reduction in the surface of the AMD-impacted wetlands. Results for Eh are presented at 10 and 20 cm to reflect changes in soil Eh with depth (Fig. 2-7). Depth is often used as a dependent variable when measuring soil Eh (Griffin and Rabenhorst, 1989; Faulkner and Patrick, 1992; Hseu and Chen, 1996).

A soil is considered reduced when all the oxygen has been removed and the NO_3^- begins to be reduced to NH_4^+. It is generally accepted that this occurs at an Eh of 200 to 250 mV (Ponnamperuma, 1972; Mitsch and Gosselink, 2000). The Eh at 10 cm in Iron Pond shows that reducing conditions were present almost the entire year at stops 2 and 4, but that at stop 5, the soil was reduced only from July to November (Fig. 2). The redox electrodes at stop 5 were placed in a hummock of sphagnum moss, which was situated above the water table, thus allowing oxidizing conditions for part of the year.

Patrick and Jugsujinda (1992) studied the sequential reduction of NO_3^-, Fe, and SO_4^{2-} in a flooded silt loam amended with rice straw at pH 6.5. They reported ferrous Fe appeared in solution only when NO_3^- had been reduced and Fe reduction began around 100 mV. They also reported SO_4^{2-} reduction occurred at –100 to –200 mV. At 10 cm in Iron Pond (Fig. 2), stops 2 and 4 had an Eh that was low enough for Fe and SO_4^{2-} reduction to occur during the warmer spring and summer months. In the winter, the higher redox potential was assumed to be due to a decline in microbial activity. The 10 cm probes were situated in the Oi layer and in this layer values for AVS were low (0.2 *u*mol g^{-1}) and values for SO_4^{2-} were high (62.3 *u*mol g^{-1}). These data indicate that despite low Eh little SO_4^{2-} from the AMD is being reduced. The values for CRS were 47.3 *u*mol g^{-1}, but this probably resulted form pyritic material washing in form the surrounding mine spoils. Iron hydroxides were observed in the Oi layer, indicating that not all iron is being reduced. This may be limiting the SO_4^{2-} reduction process. Before SO_4^{2-} is reduced most of the Fe must be reduced.

Table 3. Total sulfur (S), sulfate sulfur, acid volatile sulfides (AVS), chromium reducible sulfides (CRS), and organic sulfur for wetland soils In the Beaver Creek Watershed, WV. Organic horizons sampled by type, mineral layers sampled by depth with M1 being the uppermost layer and M4 being the lowest layer.

		Total-S			SO_4-S			AVS-S			CRS-S			Organic Sulfur-S		
		*u*mol g^{-1}dry mass														
		AMD-Impacted Wetlands														
	n*	Average	Range	Stdev	Average	Range	Stdev	Average	Range	Stdev	Average	Range	Stdev	Average	Range	Stdev
	Narrow Wetland															
Oi	3	272.4	78-489	207	0.4	0-1	0	0.6	0--1	1	31.1	0--82	44	240.4	77--406	164
Oe	1	849.4	--	--	0.9	--	--	5.3	--	--	54.0	--	--	789.3	--	--
M1	3	187.0	35--359	163	2.0	0--5	2	0.9	0--3	1	76.4	6--150	72	107.7	29--201	87
M2	3	95.7	31--129	56	0.3	0--1	0	0.3	0--3	0	32.0	3--66	32	63.1	28--101	37
M3	3	76.8	25--107	45	0.6	0--1	1	0.1	--	--	48.1	3--73	39	28.1	22--38	8
	Iron Pond															
Oi	5	237.1	54--513	173	62.3	0--160	69	0.2	0--1	1	47.3	2--200	86	127.2	32--344	126
Oe	1	36.5	--	--	0.1	--	--	0.0	--	--	0.0	--	--	36.4	--	--
M1	5	137.5	33--281	103	1.3	0--6	3	7.5	0--23	10	29.5	1--78	31	99.2	5--234	94
M2	5	50.7	25--131	45	0.7	0--3	1	0.0	--	--	3.1	0--5	2	46.9	20--127	45
M3	5	45.8	10--128	47	1.2	0--5	2	0.8	0--4	2	6.2	1--20	8	37.7	9--103	38
M4	3	19.7	16--23	4	1.2	0--3	2	0.0	--	--	2.5	2--4	1	16.0	14--16	2
	Railroad Grade															
Oi	7	185.4	83--336	86	2.5	0--7	3	18.0	0--428	46	48.2	1--215	75	116.8	0--206	70
Oe	7	268.3	105--555	154	0.1	--	--	32.3	0--215	80	58.9	2--208	84	177.0	102--283	68
Oa	5	91.6	46--151	40	0.2	0--1	0	0.0	--	--	10.2	1--45	19	81.2	46--150	41
M1	7	95.8	36--357	117	0.2	0--1	0	16.8	0--117	44	25.5	1--150	55	53.2	24--91	26
M2	7	146.0	19--666	231	1.4	0--7	2	0.1	0--1	0	13.0	1--55	19	131.4	14--164	232
M3	4	115.8	42--166	59	0.5	0--1	0	0.0	--	--	8.4	2--17	8	107.0	40--164	59
M4	2	87.4	69--105	26	5.1	0--10	7	3.0	0--6	4	31.9	1--48	22	47.4	42--53	8
		Non-Impacted Wetlands														
	ES: Alder Thicket															
Oe	2	157.3	54--165	11	0.0	--	--	0.0	--	--	15.1	7--24	12	142.2	141--143	1
Oa	2	221.8	205--239	24	0.0	--	--	0.9	0--2	1	46.2	11--81	49	174.6	156--193	26
M1	2	39.6	39--40	1	0.0	--	--	0.0	--	--	1.6	1--2	0	38.0	37--39	1
M2	2	47.1	40--54	10	0.2	--	--	0.0	--	--	2.3	2--3	1	44.6	38--51	9
M3	2	80.0	46--114	49	0.8	0--1	0	0.0	--	--	7.1	3--11	5	72.2	42--103	43
	ES: Upland Soils															
Oe	2	111.0	110--112	1	0.7	0--1	0	0.0	--	--	3.2	2--5	2	107.1	105--110	3
Oa	3	83.3	74--92	9	0.2	--	--	0.0	--	--	1.7	1--3	1	81.4	72--87	8
M1	4	20.6	21--27	3	0.0	--	--	0.0	--	--	1.1	0--3	1	19.4	20--26	3
M2	4	10.7	8--12	2	0.1	--	--	0.0	--	--	0.4	0--1	1	10.3	7--12	2
M3	4	11.0	5--14	4	0.4	0--1	0	0.0	--	--	0.1	--	--	10.5	5--14	4
M4	4	10.9	6--22	7	0.3	0--1	0	0.0	--	--	0.0	--	--	10.6	6--22	7
	ES: Bog-Marsh															
Oi	3	190.0	145--225	41	24.7	0--72	42	0.1	--	0	12.3	2--21	10	152.9	138--178	22
Oe	4	387.0	204--719	243	8.5	1--25	11	28.1	0--111	55	103.4	5--321	149	247.0	194--285	46
Oa	4	203.7	109--281	89	1.4	0--4	1	0.1	--	0	31.4	2--62	33	170.8	106--222	56
M1	2	16.4	3--30	19	0.2	--	0	0.0	--	--	0.6	0--1	1	15.7	2--29	19
M2	1	16.7	--	--	0.0	--	--	0.0	--	--	0.8	--	--	15.8	--	--
M3	1	21.7	--	--	0.1	--	--	0.0	--	--	0.3	--	--	21.3	--	--

* n - is the numebr of sampling locations that contained that layer type

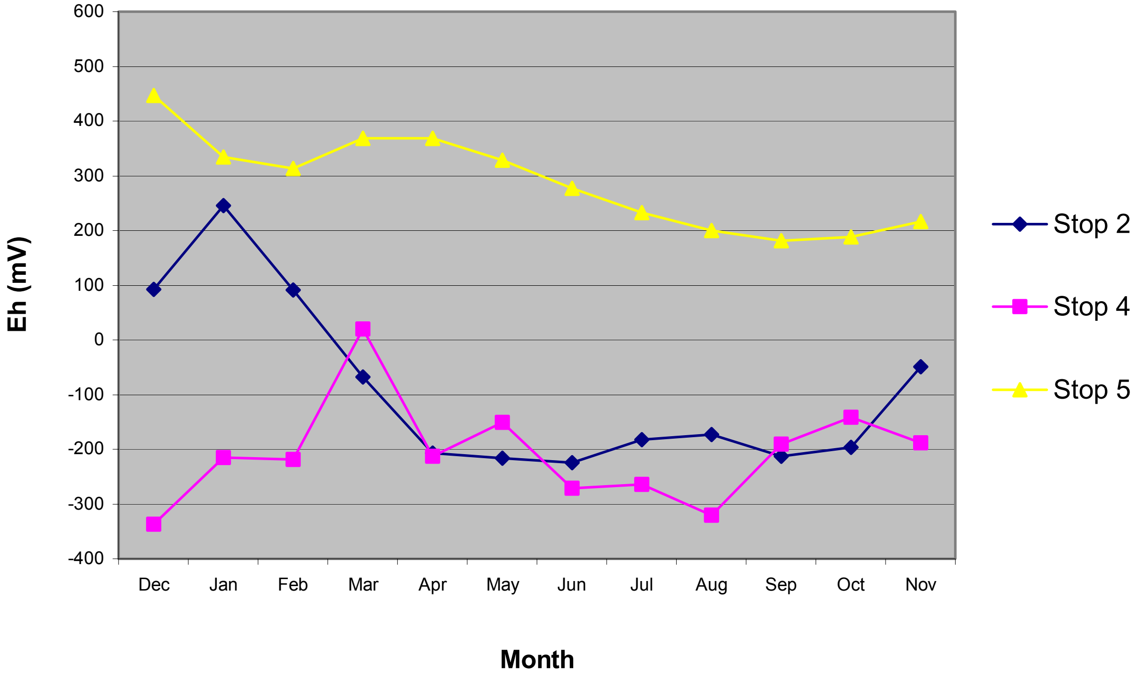

Figure 2. Redox Potential (Eh) adjusted to pH 7 at 10 cm in three sampling locations in Iron Pond wetland of the Beaver Creek Watershed, WV.

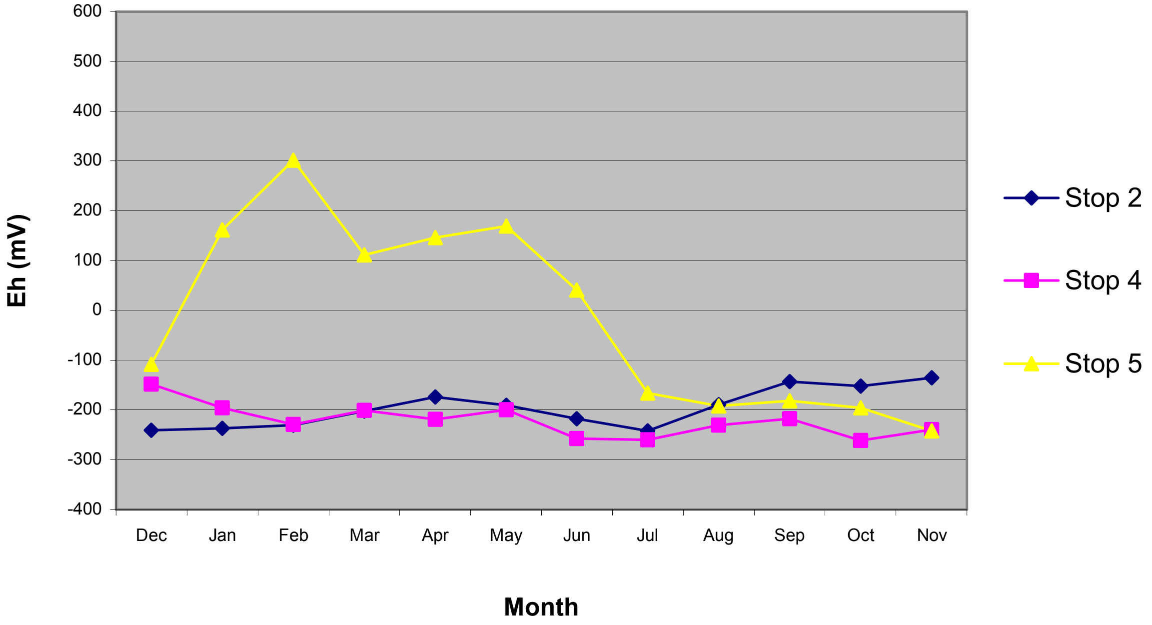

Figure 3. Redox Potential (Eh) adjusted to pH 7 at 20 cm in three sampling locations in Iron Pond wetland of the Beaver Creek Watershed, WV.

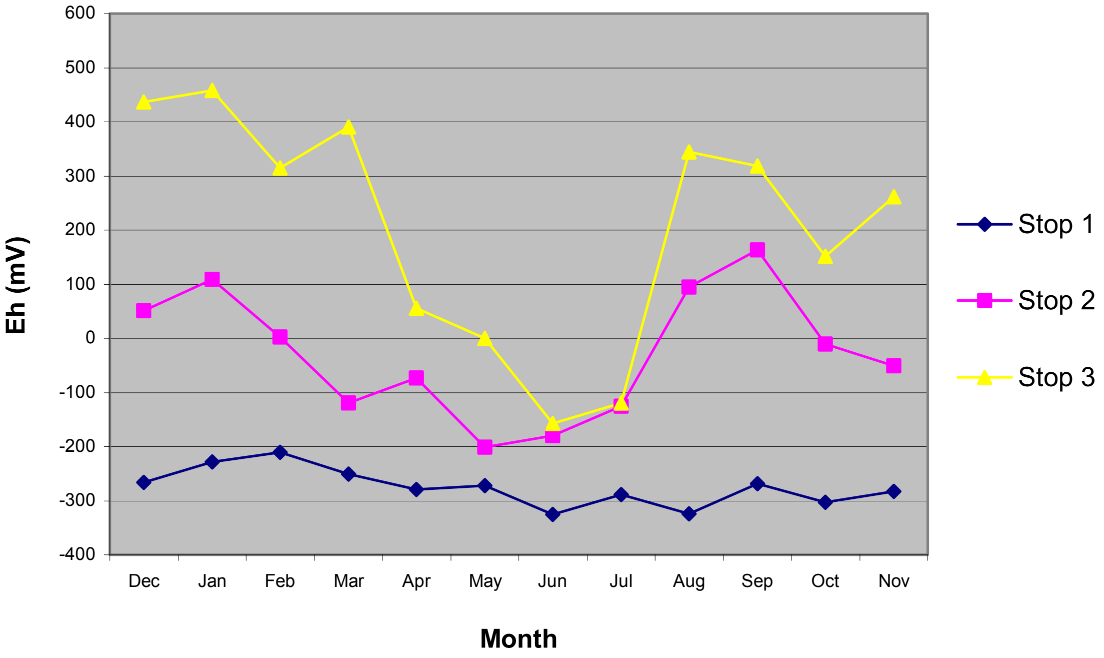

Figure 4. Redox Potential (Eh) adjusted to pH 7 at 10 cm in three sampling locations in Railroad Grade wetland of the Beaver Creek Watershed, WV.

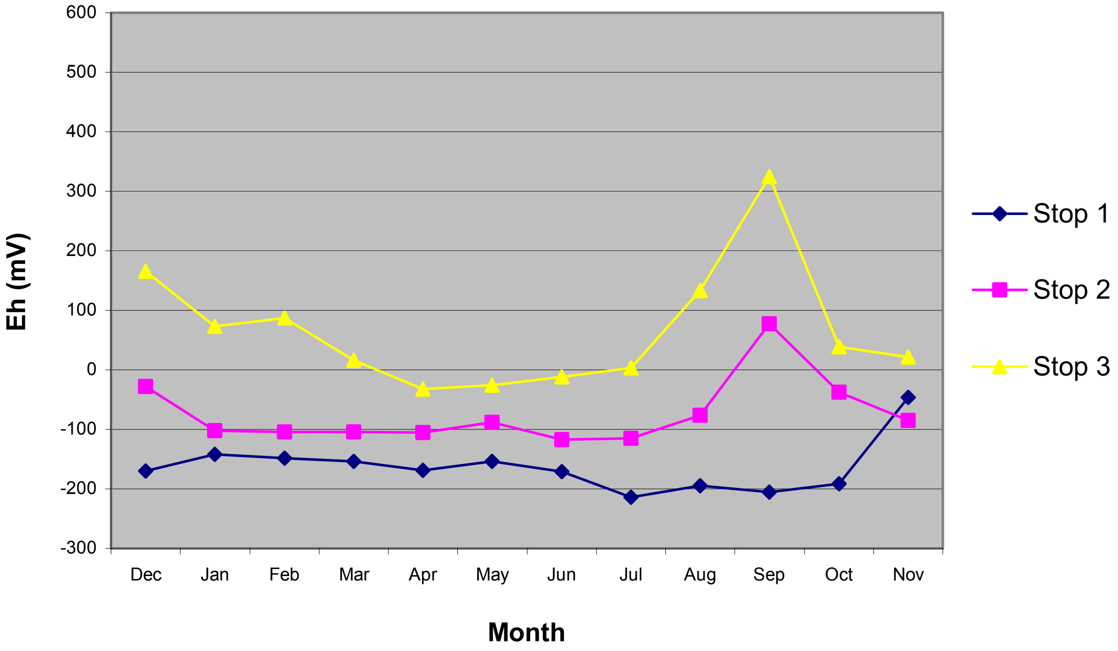

Figure 5. Redox Potential (Eh) adjusted to pH 7 at 20 cm in three sampling locations in Railroad Grade wetland of the Beaver Creek Watershed, WV.

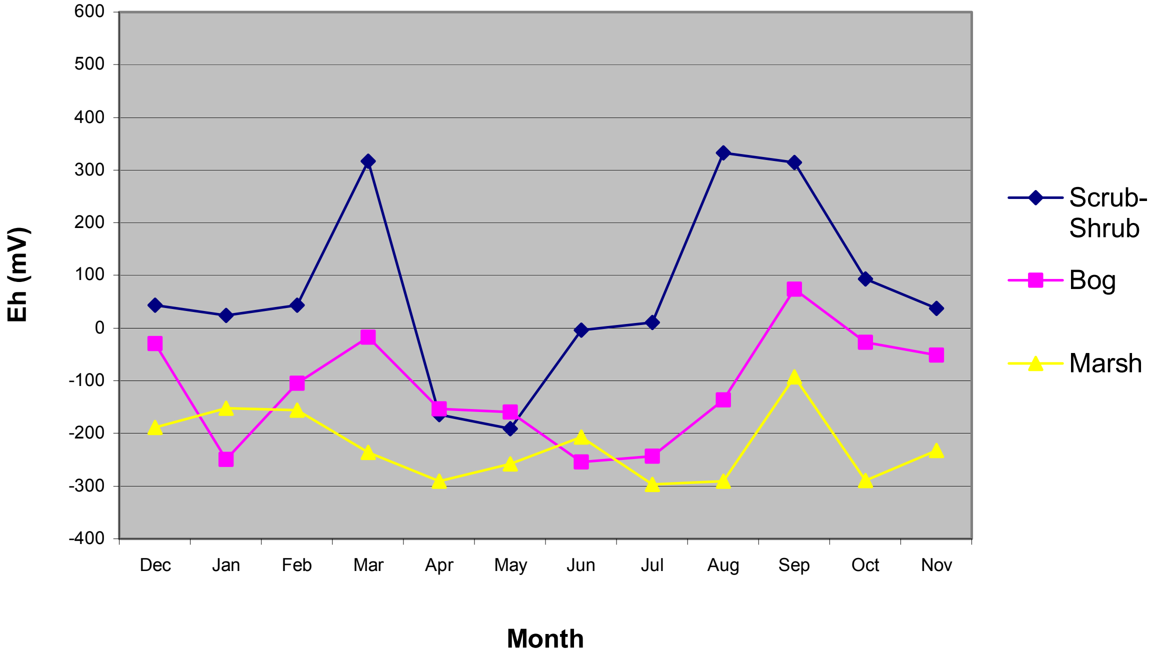

Figure 6. Redox Potential (Eh) adjusted to pH 7 at 10 cm in three sampling locations in Elder Swamp of the Beaver Creek Watershed WV.

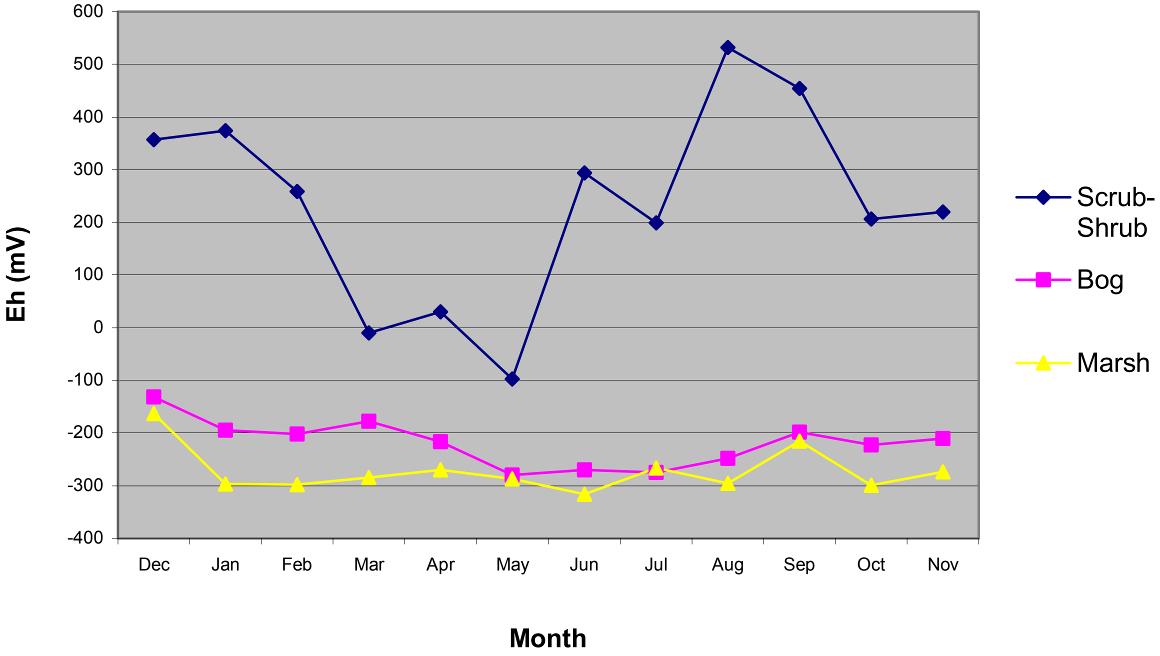

Figure 7. Redox Potential (Eh) adjusted to pH 7 at 20 cm in three sampling locations in Elder Swamp of the Beaver Creek Watershed, WV.

At 20 cm in Iron Pond, stops 2 and 4 are consistently reduced around –200 mV (Fig.3). Stop 5 had a fluctuating Eh but was still reduced most of the year (Fig. 3). These data indicate that conditions are present in stops 2 and 4 that would favor Fe and SO_4^{2-} reduction at the 20-cm depth in Iron Pond. The M1 layer at stops 2 and 4 begins at 23 and 21 cm, respectively, so redox values at 20 cm may reflect conditions in these layers. High total S content was observed in the Oi and M1 layers at these two stops. Little AVS were detected in the Oi layer, but 7.5 *u*mol g^{-1} AVS were detected in the M1 layer indicating SO_4^{2-} reduction was occurring.

In Railroad Grade, the redox electrodes were located on a transect that started at the AMD-impacted stop 1 and went slightly upslope towards non-impacted stops 2 and 3. The Eh data at both 10 and 20 cm reflected differences in hydrology along the transect, with the wetter stop 1 having a lower Eh at both 10 and 20 cm than the drier stops 2 and 3 (Fig. 4 and 5). This is similar to a trend reported by Faulkner et al. (1989) for a transect that spanned from a wetland site to a non-wetland site.

Throughout the year, in Railroad Grade, Eh at both 10 and 20 cm, at stop 1, was low enough for Fe and SO_4^{2-} to be reduced. Very high levels of S in the form of AVS and CRS were detected at stop 1 and no SO_4^{2-} was detected at 10 or 20 cm at this sampling location. Carbon is a necessary energy source for microbial reduction of S (Ponnamperuma, 1972), and C levels were above 6.6 % at these two depths. The low Eh, in conjunction with the C content, high AVS, and low SO_4^{2-} indicate that SO_4^{2-} entering the soil from the AMD is readily being reduced at this sampling location.

In Railroad Grade, stop 2 had reducing conditions present all year long at both 10 and 20 cm, and stop 3 was reduced at 20 cm most of the year (Fig. 4 and 5). At 10 cm, stop 3 was subjected to periods of reduction from April to July and was oxidized the remainder of the year (Fig. 4). Both stops 2 and 3 were not receiving AMD, and so SO_4^{2-} reduction was not of much concern at these two sampling locations.

The Eh graphs for Elder Swamp show a variable Eh for the soil in the scrub-shrub wetland at both 10 and 20 cm (Fig. 6 and 7). The Eh is lower at 10 cm than at 20 cm during most of the sampling period. At the 10-cm depth the probes are in a high-carbon A horizon and at 20 cm the probes are situated in a low-carbon B horizon. Carbon in the A horizon is providing an energy source to microorganisms, which increases the reduction rate. Less C is available in the B horizon, therefore the reduction process was inhibited. The Eh diagram for the scrub-shrub wetland indicates a transitional area where the water table fluctuates rapidly near the surface. The diagram has a degree of variability similar to that reported by Faulkner et al. (1989) for their transitional wetland, where the water table was shown to fluctuate at 30 cm.

In the bog and the marsh, at 20 cm, the Eh was consistently below -100 mV throughout the year, indicating that Fe should be reduced (Fig. 7). The bog and the marsh had conditions that favor SO_4^{2-}sulfate reduction at this depth. At 10 cm, the bog and marsh soils showed somewhat more fluctuation than at 20 cm, but the Eh was never above 100 mV (Fig. 6). The bog and marsh soils are organic soils so reducing conditions that favor organic matter accumulation were expected to prevail throughout most of the year.

Sulfur

Average total S was highest in the Oe layer of Narrow Wetland (849.4 *u*mol g^{-1}) and lowest in the M2 layer of the upland soils (10.7 *u*mol g^{-1}) (Table 3). When comparing the Oi layers in all of the wetlands, we observed that the average total S was generally higher in the three impacted wetlands than in the bog-marsh in Elder Swamp (Table 3). The average total S of the Oi layer in Railroad Grade (185.4 *u*mol g^{-1}) was actually slightly less then what we observed in the bog-marsh (190.0 *u*mol g^{-1}), but the high end values of the range for the Oi layer in Railroad Grade were observed in the AMD-impacted portion of this wetland (Table 3). Furthermore, average total S in the bog-marsh wetland may be elevated due to the presence of a nearby coal refuse pile. The drainage from the coal refuse pile flows into the marsh area of the wetland after it is treated to raise the pH and precipitate the iron as iron oxides into settling ponds. The SO_4^{2-} that remains in this drainage allows S to accumulate. Higher S values were observed in the individual sampling spots located in the marsh area than in the bog area.

Our average total S values for all AMD-impacted Oi layers were much higher than the 108 *u*mol g^{-1} Wieder and Lang (1986) reported for Tub Run Bog, and were similar to those reported by Griffin and Rabenhorst (1989) for the surface layer of a tidal marsh soil. Tidal marsh soils receive sulfate-rich seawater, so our data indicate that the surface Oi layers of our wetlands are receiving and retaining additional S from the inflowing AMD.

Sulfur was generally higher in the mineral layers of the AMD-impacted wetlands than in the non-impacted wetlands (Table 3). The M1 layers in all three of the AMD-impacted wetlands had average total S values similar to those reported by Wieder and Lang (1986) for the surface layer in Tub Run Bog. This indicates that additional S from the AMD is entering these layers and being retained.

Average SO_4^{2-}-S was highest in the Oi layer of Iron Pond (62.3 *u*mol g^{-1}) and SO_4^{2-} constituted 26.3 % of the total S pool in this layer (Tables 3 and 4). The Oi layer in the bog-marsh had the next highest SO_4^{2-} concentration (24.7 *u*mol g^{-1}), and it constituted 13.0 % of the total SO_4^{2-} pool (Tables 3 and 4). In Tub Run Bog, Wieder and Lang (1986) observed that only 5.4 % of the S was in the SO_4^{2-} form. Laudon's (1989) data for a constructed wetland designed to treat AMD showed that 2.9 to 25 % of the total S was in the SO_4^{2-} form. Our data indicate that SO_4^{2-} from the AMD is being retained in Iron Pond, and SO_4^{2-} from the nearby coal refuse pile is being retained in the bog-marsh wetland. Furthermore, not all SO_4^{2-} entering these wetlands is reduced.

The Oi layer in Iron Pond was saturated with iron hydroxides, and almost no AVS was detected in this layer. Since AVS indicates SO_4^{2-} reduction, the abundance of iron hydroxides along with the lack of AVS indicates that SO_4^{2-} reduction was not occurring in this layer. For SO_4^{2-} to be reduced, the Fe, in the ferric form, must first be reduced to ferrous Fe. Therefore, the abundance of iron hydroxides indicated that Fe had not been completely reduced and SO_4^{2-} reduction was not readily occurring.

The highest average AVS value was found in the Oe layer of Railroad Grade (32.3 *u*mol g^{-1}) and the next highest average AVS value was in the Oe layer of Bog-Marsh (28.1 *u*mol g^{-1}) (Table 3). The AMD-impacted portion of Railroad Grade had the highest average AVS in the Oi, Oe, and M1 layers of any wetland (Table 3). No AVS was detected in the non-impacted section of this wetland. The highest AVS value observed at any one individual sampling location in any wetland was 428 *u*mol g^{-1} in the Oi layer of an AMD-impacted sampling location in Railroad Grade. This was about 4 times greater then any value Laudon (1989) reported and 138 times greater than the 3.1 *u*mol g^{-1} that Wieder and Lang (1986) reported for Tub Run Bog surface layer. These data, along with the lack of SO_4^{2-}, indicate that SO_4^{2-} reduction was occurring in the AMD-impacted surface organic layers of Railroad Grade. In Narrow Wetland the Oe layer had 5.3 *u*mol g^{-1} AVS, which was similar to values reported by Wieder and Lang (1986). The lower average value for AVS indicated that little sulfur had been recently reduced in Narrow Wetland.

Table 4. Percentage of total sulfur that each form of sulfur constitutes in wetland soils in the Beaver Creek Watershed, WV. Organic horizons sampled by type, mineral layers sampled by depth.

		SO4-S	AVS	CRS	Organic-S
		% of total S			
		AMD-Impacted Wetlands			
	n*				
Narrow Wetland					
Oi	3	0.1	0.2	11.4	88.2
Oe	1	0.1	0.6	6.4	92.9
M1	3	1.1	0.5	40.9	57.6
M2	3	0.3	0.3	33.4	66.0
M3	3	0.8	0.1	62.6	36.5
Iron Pond					
Oi	5	26.3	0.1	20.0	53.7
Oe	1	0.3	0.0	0.0	99.7
M1	5	0.9	5.4	21.5	72.1
M2	5	1.3	0.1	6.0	92.6
M3	5	2.6	1.8	13.5	82.2
M4	3	5.8	0.2	12.7	81.2
Railroad Grade					
Oi	7	1.3	9.7	26.0	63.0
Oe	7	0.0	12.1	22.0	66.0
Oa	5	0.3	0.0	11.1	88.6
M1	7	0.2	17.6	26.6	55.6
M2	7	1.0	0.1	8.9	90.0
M3	4	0.4	0.0	7.2	92.4
M4	2	5.9	3.5	36.5	54.2
		Non-Impacted Wetlands			
ES: Alder Thicket					
Oe	2	0.0	0.0	9.6	90.4
Oa	2	0.0	0.4	20.8	78.7
M1	2	0.0	0.0	4.0	96.0
M2	2	0.4	0.0	4.8	94.8
M3	2	0.9	0.0	8.9	90.2
ES: Upland Soils					
Oe	2	0.6	0.0	2.9	96.5
Oa	3	0.2	0.0	2.1	97.7
M1	4	0.1	0.0	5.5	94.4
M2	4	0.9	0.0	3.5	95.6
M3	4	3.9	0.0	1.2	95.0
M4	4	3.2	0.0	0.1	96.7
ES: Bog-Marsh					
Oi	3	13.0	0.0	6.5	80.5
Oe	4	2.2	7.3	26.7	63.8
Oa	4	0.7	0.1	15.4	83.8
M1	2	1.1	0.0	3.5	95.4
M2	1	0.3	0.0	4.9	94.8
M3	1	0.5	0.0	1.4	98.1

* n - is the numebr of wetland sites that contained that layer type

No AVS was detected in the upland soils and very little was observed in the alder thicket (Table 3). Except for the M1 layers in all AMD-impacted wetlands, very little AVS was detected in any of the mineral layers. The M1 layers in all AMD-impacted wetlands had an average AVS of 0.9 to 16.8 *u*mol g^{-1}. This indicates that some S was being reduced and retained in the first mineral layer.

The upland soils and the alder thicket had the lowest amount of CRS detected (Table 3). The highest CRS value observed was in the Oe layer of the bog-marsh wetland (103.4 *u*mol g^{-1}) and this value was similar to the 131.0 *u*mol g^{-1} of CRS reported by Laudon (1989). At least 2.5 *u*mol g^{-1} of CRS was detected in all AMD-impacted wetland mineral and organic layers, indicating that at least some S reduction has occurred in all of the AMD-impacted wetlands (Table 3). In the bog-marsh wetland, less than 1.0 *u*mol $g^{-1}$of CRS was observed in any mineral layer. This is related to the lack of Fe in these layers, which is required to form iron sulfides. The average values for CRS in the organic layers of the AMD-impacted wetlands (except for the Oe layer in Iron Pond) were much higher than the 15.1 *u*mol g^{-1} that Wieder and Lang (1986) reported for Tub Run Bog. Yet, our total S values were higher than the values reported for Tub Run Bog and the percentage of the total S that our CRS constituted was similar to the 14 % they reported (Table 4)

Wieder and Lang (1986) reported that 75 % of the sulfur in the surface layer of AMD-impacted Tub Run Bog was in the organic form and 85 to 88 % of the S in their three non-impacted wetlands was in the organic form. Table 4 shows that for all organic layers in the alder thicket and the upland soils, 78.7 to 97.7 % of the S was in the organic form. If we include the bog-marsh with the AMD-impacted wetlands, we observed that for all organic layers of the AMD-impacted wetlands, 53.7 to 99.7 % of the S was in the organic form. The AMD-impacted wetlands tended to have more S in the form of CRS, while in Railroad Grade and Iron Pond, the S was likely to be in the AVS and or SO_4^{2-} forms.

Summary

Wetlands impacted by AMD tended to have lower carbon content and higher Fe concentrations in the organic layers. These findings are similar to those Wieder and Lang (1986) reported for the AMD-impacted Tub Run Bog. They attributed the reduced organic matter content to translocation of inorganic particles from the nearby mine spoils, and the increased iron content to the inflowing AMD. We believe that these same processes are occurring in our AMD-impacted wetlands.

Redox potential in Iron Pond and Railroad Grade is low enough to indicate that Fe and S from the inflowing AMD could be reduced in these wetlands. In Railroad Grade, the organic layers had high values for total S, AVS, and CRS along with low SO_4^{2-} values, which indicate that S is being reduced in this wetland. In Iron Pond almost no AVS was detected in the Oi layer and SO_4^{2-} constituted 26.3 % of the total S pool there. This was attributed to the abundance of oxidized ferric iron, which must first be reduced before the SO_4^{2-} could be reduced to the sulfide form. Sulfate reduction does not appear to be readily occurring in the Oi layer of Iron Pond. In Iron Pond 47.3 *u*mol g^{-1} of CRS was detected in the Oi layer, but this may be from the translocation of pyritic material from the adjacent mine spoils, not from the formation of CRS via reduction of SO_4.

Sulfate in drainage from a nearby coal refuse pile appear to be entering the soils in the bog-marsh wetland. Our data indicate that most of the SO_4^{2-} in this wetland was being retained and reduced in the marsh portion of the wetland, which received the majority of the AMD. The redox potential for the marsh area indicated that conditions that favor sulfate reduction are present at both 10 and 20 cm are present.

In general, the AMD-impacted wetlands tended to have more total S, AVS, CRS, and SO_4^{2-} than the non-impacted wetlands. In non-impacted wetlands S was more likely to be in the organic form, and almost no sulfur was in the AVS form. Of the organic layers in all non-impacted wetlands, SO_4^{2-} or AVS sulfur was found to constitute more than 0.6 % of the total S pool in only the Oi and Oe layers of the bog-marsh, which receives AMD.

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