

PROCESSES OF ALKALINITY ADDITION TO
PASSIVE WETLAND SYSTEMS NEAR
GREAT FALLS, MT

BY:

Diana L. Allen

A professional paper submitted in partial fulfillment of the
requirements for the degree
of
Master of Science
in
Land Rehabilitation

MONTANA STATE UNIVERSITY
Bozeman, Montana
May 1998

P378
AL533

ii

Approval

of a professional paper submitted by

Diana Allen

This professional paper has been read by each member of the professional paper committee and has been found to be satisfactory regarding content, English usage, format, citations, bibliographic style, and consistency, and is ready for submission to the College of Graduate Studies.

April 21, 1998
Date

Douglas J. Dollhopf
Chairperson, Graduate Committee

Approved for the Major Department

4-21-98
Date

Peter Buehner
Head, Major Department

Approved for the College of Graduate Studies

5/5/98
Date

Joseph J. Felsch
Graduate Dean

STATEMENT OF PERMISSION TO USE

In presenting this paper in partial fulfillment of the requirements for a master's degree at Montana State University, I agree that the Library shall make it available to borrowers under rules of the Library.

If I have indicated my intention to copyright this paper by including a copyright notice page, copying is allowable only for scholarly purposes, consistent with "fair use" as prescribed in the U.S. Copyright Law. Requests for permission for extended quotation from or reproduction of this paper in whole or in parts may be granted only by the copyright holder.

Signature

Diana L. Allen

Date

4-30-98

TABLE OF CONTENTS

INTRODUCTION.....	1
REVIEW OF STUDY WETLANDS.....	1
French Coulee Wetland.....	2
Centerville "C" Wetland.....	10
Stockett SE Site.....	17
Northwest Interceptor Site.....	17
Southwest Interceptor Site.....	19
WETLAND PROCESSES.....	22
Aerobic Wetland Processes.....	22
Anaerobic Wetland Processes.....	25
Acid Mine Drainage Chemistry.....	29
Sizing Study Wetlands.....	30
POTENTIAL TREATMENT METHODS TO ENHANCE WETLAND PERFORMANCE.....	35
Anoxic Limestone Drains.....	35
French Coulee and Centerville	39
Stockett System.....	41
Open Limestone Channels.....	42
French Coulee Wetland.....	46
Centerville Wetland.....	47
Stockett System.....	47
Sustained Alkalinity Generating System.....	48
French Coulee Wetland.....	51
Centerville Wetland.....	51
Algae-Microbial Removal of Manganese and Iron.....	52
Reverse Alkalinity Production Systems.....	53
Limestone Ponds.....	55
CONCLUSION.....	57
LITERATURE CITED.....	59

LIST OF TABLES

Table	Page
1. Drainage chemistry at the Study Sites.....	9
2. Possible Cell mode combinations-Centerville	12
3. C:N Comparison of Centerville Cells to a natural wetland.....	17
4. A comparison of aerobic and anerobic wetlands.....	23
5. Eastern U.S. Acid Mine Drainage Chemistry.....	31
6. Comparison of chemical parameters of the oxized and reduced areas of French Coulee and Centerville substrates.....	41
7. Construction criteria for open limestone channels....	44

LIST OF FIGURES

Figure	Page
1. Planview of French Coulee wetland site.....	3
2. Planview of Centerville wetland site.....	11
3. Planview of Stockett site.....	18
4. Sustained alkalinity producing system.....	50
5. Schematic diagrams of passive treatment systems.....	54

ABSTRACT

Acid mine drainage is a by-product of coal mines across the United States. The State of Montana Abandoned Mine Reclamation Bureau constructed three passive wetland systems to treat acid mine drainage near Great Falls, Montana. The passive wetland systems have not adequately treated acid mine drainage. The addition of alkalinity to the passive wetland systems may enhance their performance. Processes to add alkalinity include: anoxic limestone drains, limestone ponds, open limestone drains, successive alkalinity generating systems, reverse alkalinity generating systems, and algae mat consortiums. The purpose of this study is to determine possible causes for the failures of the wetlands to fully treat acid mine drainage and possible solutions to treat acid mine drainage.

INTRODUCTION

Acid mine drainage (AMD) is a common problem associated with coal and metal mines and is a byproduct of pyritic mineral dissolution. Near Belt, Montana large abandoned underground coal mines of the Great Falls coal fields produce acid mine drainage through seeps which discharge to the natural water system. To remediate the AMD the State of Montana Abandoned Mine Reclamation Bureau (AMRB) constructed three passive wetland treatment systems. Since their construction, the performance by these systems has been less than expected. The AMRB engaged Montana State University Reclamation Research Unit to review the Belt passive wetland treatment systems and to explore alternate methods to improve their performance. The objective of this paper is to review the latest literature on the addition of alkalinity to wetland systems and to make recommendations to improve the ability of these wetlands to treat AMD.

REVIEW OF STUDY WETLANDS

The State of Montana Abandoned Mine Bureau constructed three wetlands to treat acid mine drainage originating from underground coal mines near Great Falls, Montana. Since the beginning of operation the wetlands have not functioned as

designed. Physical flow problems are present in all three. Winter freezing produces thick sheets of ice on pond surfaces and uncontrolled flow patterns result in loss of containment and limited opportunity for proper contact between the water and the substrate. Inorganic precipitates and bacterial slime foul the buried pipe systems causing continuing plugging problems and restrict flow through the substrate. At the Stockett and Centerville sites failure or plugging of anoxic limestone drains (ALD) has resulted in a significant loss of chemical pretreatment since the beginning of operation.

French Coulee Wetland

The French Coulee site is near Belt, Montana. The site was constructed between fall 1990 and spring 1991 and consists of three cells in series (Figure 1). It receives 21.2 to 63.4 Lpm (5 to 15 gpm) of acid mine drainage. The system is designed in either an upflow or downflow or in a crossflow (surface) system. The size of the treatment area is 0.48 hectares (1.3 acres). The substrate is a mixture of wood waste and municipal sewage and one layer of aged cow manure above the gravel layer (Schafer and Associates 1994).

The wetland was flooded with uncontaminated stream water prior to introducing acid mine drainage. Treatment of acid mine drainage began on July 2, 1991 in the upflow mode.

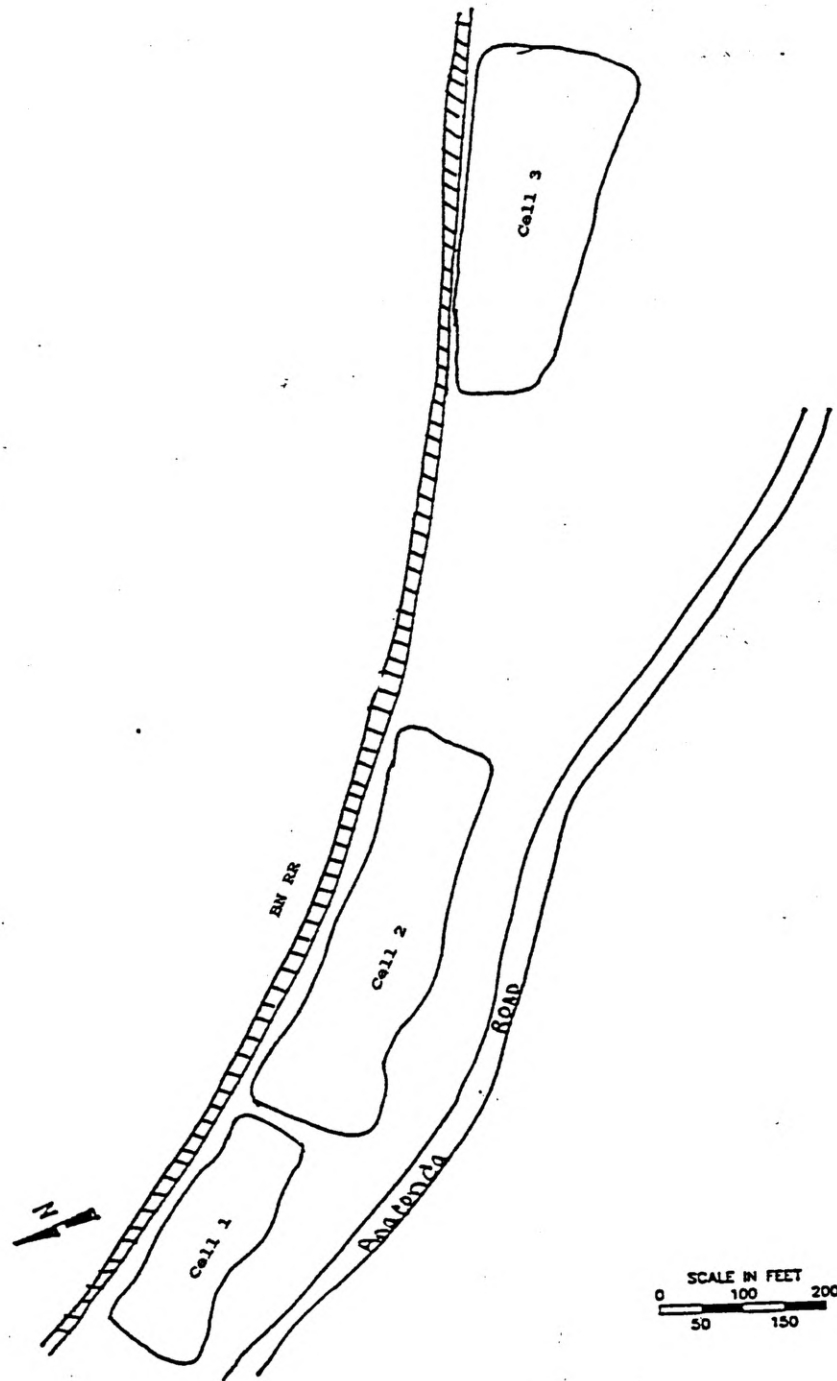


Figure 1. Planview of French Coulee site (Adapted from Schafer and Associates 1994).

to minimize contact of AMD with oxygen and to help maintain reducing conditions. Site visits in August and October 1991, showed AMD was breaking through, meaning that all the acid mine drainage was not adequately treated, from the final treatment cell (Schafer and Associates 1994). A clogged screen was cleaned and this appeared to allow the system to function again. However, the screen had to be cleaned every few weeks to correct the problem. In January, the system short circuited and iron staining was observed in a small stream down gradient from the wetland indicating that the system was not treating AMD. The system was shut down and the AMD was diverted back into the original collection ditch. In the summer of 1992, a valve was installed in the main pipeline to facilitate water diversions from the wetland to a collection ditch as necessary. Flow was reestablished in August in a downflow mode to help solve the plugging of screen and pipes. Again in winter of 1992 ice sheets developed, the system was shut down and water diverted back to the original channel. During early 1993, it was determined that partial plugging of the substrate distribution piping was preventing adequate treatment of AMD. In Cells 1 and 2, a portion of the flow was passing through the substrate. The exit weirs, when set at maximum height, could not provide sufficient head to produce flow through the media. A sewer cleaning service was hired to clean out the main pipeline and distribution

pipng in the cells in October, 1993. Due to time constraints Cell 3 wasn't completely cleaned. After the cleaning, Cell 1 was performing better but 40% of total flow was in crossflow or in a surface flow mode (Schafer and Associates 1994). Cell 2 was totally in a downflow mode. Cell 2 effluent had a white milk appearance and white coating which may indicate aluminum was precipitating out of the AMD. The fouling of the perforated piping was due primarily to sediment and consolidated gelatinous sediment globules less than one half inch in diameter. The sediment became darker from Cell 1 to Cell 3 which would indicate reducing conditions. It was not determined if the globules were from bacterial activity or inorganic precipitates. However, bacteria fouling was found in the eight inch main.

Total acidity and metals reduction in the French Coulee wetland effluent was only about one-fourth of what is required to maintain near neutral pH levels. One half of the reduction in acidity was due to inherent natural alkalinity present in the substrate. Cell 1 seems to be depleted of natural alkalinity due to the lack of treatment of AMD. Cell 2 is providing alkalinity in the downflow mode. Cell 3 was in a crossflow mode and no alkalinity was likely from the substrate. The wetland was having trouble maintaining reducing conditions as indicated by variable ferrous iron content. The pH of the effluent from Cell 3 has dropped to 2.9 compared to pH 4.1 effluent from Cell 2

even though more iron was being removed in Cell 3 than Cell 2. Aluminum was not being immobilized and most aluminum precipitation is happening in Cell 2. Schafer and Associates in 1994, reported that Cell 1 alkalinity for metal reduction was from inherent natural substrate and not from other processes such as bacterial reduction but only 50% of the observed acidity reduction can be attributed to increases of Ca and Mg (Schafer and Associates 1994). The substrate in Cell 1 must not be functioning, not even treating the AMD that does make contact with the substrate. Cell 2 was still providing treatment which could have been from sulfate reducing bacteria but the low pH is suppressing bacteria activity. The bacteria activity could be concentrated around localized areas of inherent natural alkalinity provided by the substrate.

The French Coulee site was seeded in the fall of 1990. Grass species seemed to be establishing themselves. The noxious plants, spotted knapweed and thistles were present. Trees and shrubs established poorly. Cattails in Cell 1 established poorly, even in 1995 cattail establishment lacked vigor. Most cattails were lost after flooding the wetlands and the survivors have undersized seed heads and are slightly deformed. Cattails in Cell 2 were more abundant but still had a pale color and were only 8-12 inches tall. The best establishment of cattails was in Cell 3 with approximately 80% coverage.

Several problems associated with the French Coulee site prevent the wetland from performing as expected. The plugging of the distribution plumbing does not allow the wetland cells to function in the more efficient anaerobic mode by preventing AMD contact with the substrate. Because the wetland flow has been in a aerobic or crossflow mode damage to the substrate may have occurred due to the desiccation of the substrate. Since the water flows across the surface of the pond, the substrate has little contact with AMD and little chance to treat it (Wildeman et al. 1993). The substrate efficiency may be decreased because of the low pH levels of the influent to the wetland cells. According to Wildeman bacterial activity is repressed by the low pH from flooding the wetland with AMD immediately after construction. Flooding the wetland with relative clean water would allow bacteria time to establish function (Wildeman et al. 1993). Although the wetland was flooded with clean water before the introduction of AMD, it was possible that this system may have insufficient time to establish an active bacterial population. It was also possible that the substrate doesn't host a sulfate reduction bacteria, by inoculating the substrate during construction of the wetland. At pH levels below five there is a repression in bacterial activity in substrates (Wildeman et al. 1993). The addition of alkalinity would improve the capacity of the substrate bacteria to treat acid mine

drainage. It would also increase metal complexation due to precipitation which is more efficient at pH 4 and above (Wildeman et al. 1993). The mechanism of exchange of metals into organic matter will not occur until the pH is above three. Since the pH in the wetland was below three, we can assume that these processes are not occurring efficiently. During the winter months bacteria activity was slowed by the low winter temperatures. The cold temperatures caused large ice sheets to form on the surface of the wetlands and cause the plumbing and weirs to freeze preventing good flow through the wetlands.

At the end of the study, the French Coulee passive wetland was partially treating the acid mine drainage. Schafer and Associates (1994) reported to the State of Montana that the effluent from the wetland showed improvement in reduction of iron (-36%), aluminum (-40%), zinc (-45%), and a slight reduction in sulfates (-4%). The system was also producing alkalinity, adding calcium (547 mg/L) and magnesium (141 mg/L) to the effluent. A disturbing point was the addition of manganese to the system, 0.5 to 4.9 mg/L. Total dissolved solids content showed improvement in treatment, a reduction of 4.7%. However, there was virtually no improvement in acidity with pH at 2.9.

Table 1. A comparison of effluent from the three wetlands and water quality standards (Schafer and Associates 1994).

Component	French Coulee Wetland			Centerville C Wetland			Stockert SE Wetland (West System)		
	Influent Water	Effluent Water	% Change	Influent Water	Effluent Water	% Change	Influent Water	Effluent Water	% Change
Fe (mg/l)	555.0	354.0	-38.2	58.0	5.0	-91.4	1170.0	878.0	-25.0
Al (mg/l)	290.0	173.0	-40.3	263.0	18.7	-92.9	1200.0	1250.0	4.2
Ca (mg/l)	178.0	547.0	207.3	229.0	542.0	136.7	450.0	465.0	3.3
Mg (mg/l)	91.0	141.0	55.0	95.0	124.0	30.5	290.0	293.0	1.0
Mn (mg/l)	0.5	4.9	880.0	2.54	4.55	79.1	8.81	9.72	10.3
Zn (mg/l)	2.70	1.48	-45.2	2.84	0.18	-93.7	62.8	65.4	4.1
SO ₄ ²⁻ (mg/l)	3680	3530	-4.1	3150	2150	-31.8	10400	10600	1.9
TDS (mg/l)	5320	5070	-4.7	5020	3280	-34.7	15700	16200	3.2
pH (su)	2.4	2.9	NMF	2.7	4.4	NMF	2.6	2.6	NMF

NMF = No Meaningful Figure

Component	Drinking Water MCLs	Fresh Water Aquatic ¹ Acute/Chronic	French Coulee Wetland Effluent Water	Centerville C Wetland Effluent Water	Stockert SE Wetland Effluent Water
			Most Recent Data	Most Recent Data	Most Recent Data
Fe (mg/l)	0.3	1.0/None	354.0	5.0	937.0
Al (mg/l)	None	None	173.0	18.7	1050.0
Ca (mg/l)	None	None	547.0	542.0	435.0
Mg (mg/l)	None	None	141.0	124.0	258.0
Mn (mg/l)	0.05	None	4.9	4.6	10.6
Zn (mg/l)	5.0	320/047	1.48	0.2	65.5
SO ₄ ²⁻ (mg/l)	250	None	3530	2150	9190
TDS (mg/l)	500 ¹	None	5070	3280	13800
pH (su)	5 to 9	None/6.5 to 9.0	2.9	4.4	2.7

¹ The drinking water sets the TDS requirement based on total sulfate plus chloride. ² Calculated @ 100 mg/L hardness.

Centerville C Wetland

Centerville "C" wetland is located near the park in the small town of Centerville, southeast of Great Falls.

Centerville "C" wetland is 0.37 hectares (1.0 acres) and receives 21.1 to 42.2 Lpm (5 to 10 gpm) of AMD (Figure 2). An anoxic limestone drain (ALD) was constructed to provide pretreatment of AMD. The wetland operated in a upflow mode and water was conveyed to each cell through a short section of perforated pipe buried in a gravel subbase. There was no distribution piping in the cells. The substrate was a mushroom compost and pulverized limestone (Schafer and Associates 1994).

The Centerville design consisted of interceptor drains, anoxic limestone trench, and a three cell wetland (Peccia and Associates 1992). Acid mine drainage seeps were collected by interceptor pipes which were joined by non acidic surface water from a collection basin to be discharged to Cell 1. The water collected in the interceptor drains was discharged into Service Hole #1. There the AMD was directed into the anoxic limestone drain (ALD) or the water could be bypassed around the ALD. The surface water from the collection basin also flowed into Service hole #1 and was directed onto Service Hole #2. The surface water cannot be directed to flow into the ALD.

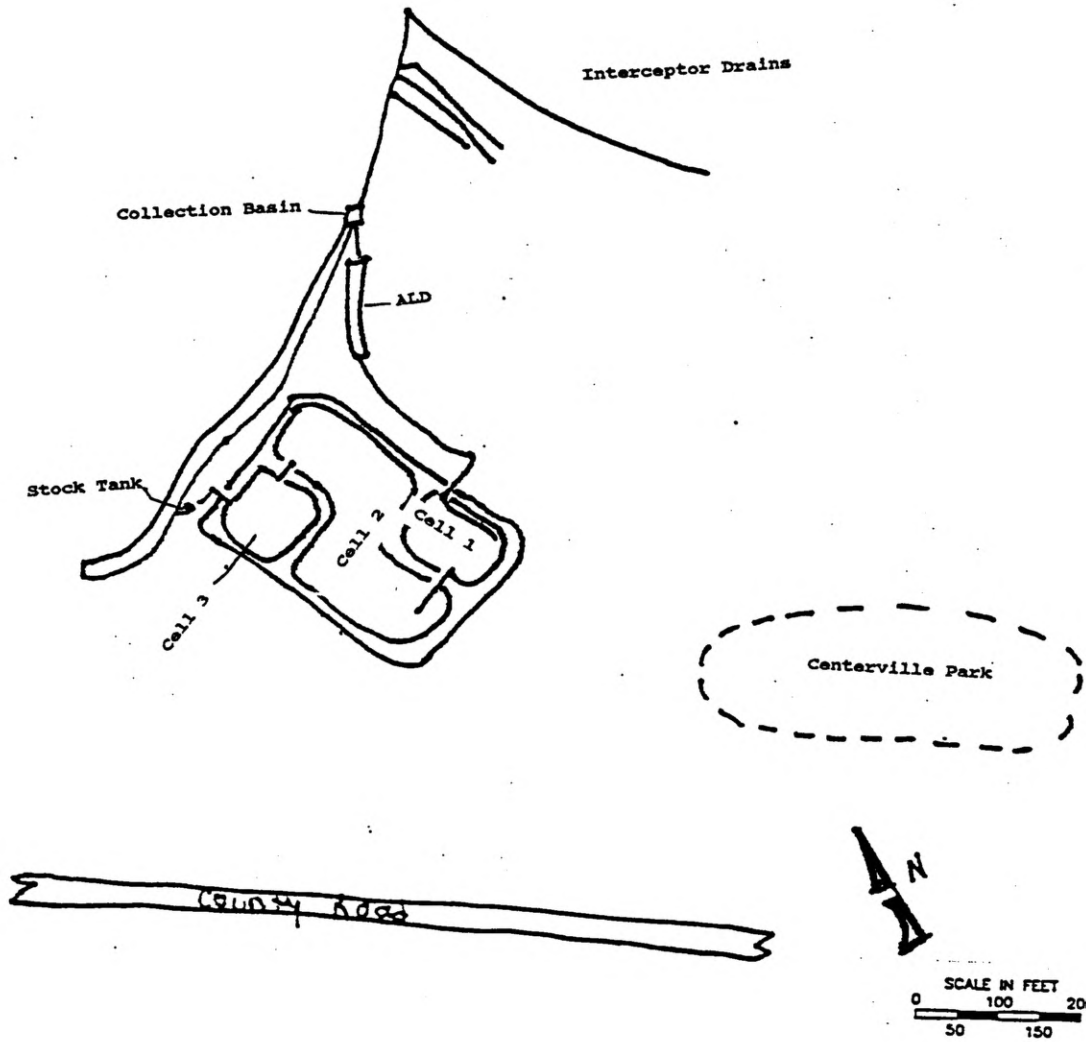


Figure 2. Planview of Centerville wetland site. Adapted Schafer and Associates 1994.

Service Hole #2 directed both inflow sources to wetland Cell 1 or bypassed both sources into the north drainage. Effluent water leaving wetland Cell 3 can be routed to flow into a livestock tank for use in watering stock or bypassed into the north natural drainage ditch.

Each Centerville "C" wetland cell may flow in a surface or subsurface mode. Peccia and Associates developed different scenarios to test the effectiveness of each cell under different operating conditions and combinations (Peccia and Associates 1992). Treating AMD in different

Table 2. Possible Cell Mode Combinations-Centerville "C".

Possible Cell Mode Combinations-Centerville "C"				
Scenario	Cell 1	Cell 2	Cell 3	ALD
1	Surface	Surface	Surface	Off
2	Subsurface	Surface	Surface	Off
3	Surface	Surface	Subsurface	On
4	Subsurface	Subsurface	Subsurface	On & Off
5	Surface	Surface	Surface	On & Off

combinations and observing the results would indicate the best option for maximizing treatment of AMD. The wetland designers felt the ALD was to be used judiciously and reserved for use when the efficiency of the buffering capacity in the substrate diminished (Peccia and Associates 1992). Scenario 4 was developed to test the wetland in reducing

conditions with the ALD on and off. Scenario 5 was designed to test oxidizing conditions with the ALD on and off. The testing of each scenarios should be done for one year or at least for one residence time of water in the wetland.

In the spring of 1991, the wetland was designed to accommodate the largest available area at the site by using iron loading rate. It was found that designing the size of the wetland on iron loading rather than acid loading lead to the design of a wetland to small in size to treat the AMD (Peccia and Associates 1993). The new recommendation for size of the wetland, based on acidity, was limited by the area available at the site and wetland size was to small based on acidity loading.

The substrate in Centerville Cell 1 consisted of a filter underlayer of gravel with compost growth medium (compost peat) above the gravel. Two types of EKO Compost peat, a commercial product, were used: standard peat, peat DCW-peat with added dried poultry waste, and peat OM-cattle manure and straw. Peat OM consisted of cattle manure and straw composted on the site. The peat was mixed with limestone aggregate, 25% by weight. Cell 1 was built with EKO compost with limestone aggregate for the substrate. Due to the lack of EKO compost, Cell 2 substrate was constructed using a mixture of cow manure and straw with crushed limestone. Cell 3 contained peat DCW (EKO peat and dried

poultry waste), a mixture designed to facilitate anaerobic processes.

The wetland functioned from July 21, 1992 till November 19, 1992 when a failure to treat AMD occurred in the ALD. This failure was due to plugging of the ALD. No AMD reached the wetland. Drainage was flowing across the surface because of plugging in the ALD and in the buried inlet pipe, and because of winter ice sheets (Schafer and Associates 1994). In the summer of 1993, the State AMRB tried to resume flow through the ALD but the attempt was unsuccessful and the ALD was bypassed. Schafer and Associates (1994) theorized that the alkalinity (Ca and Mg) present in the effluent was from the ALD, while it was working, then the alkalinity was derived from the substrate. Evidence of wetland treatment of AMD was the reduction of sulfate, aluminum and iron from cell to cell. In the crossflow mode (surface flow) oxidized iron was precipitated and residual iron in the effluent was present predominantly as ferric iron. High levels of iron and aluminum were being removed but in July 1993, Cell 1 had a breakthrough of AMD with a pH of 2.9. Soluble iron was not being precipitated and it seemed that the iron precipitate was being dissolved. It was theorized that the cause of the failure was due to depletion of alkalinity in Cell 1. This situation may occur in the other cells. However, a substrate sample was taken from Cell 1 during a field visit in February, 1995. The

sample showed limestone fragments were still present in the substrate and that were not armored. Due to the presence of unarmored limestone fragments, it is unlikely that the cell was depleted of alkalinity but it may have been overwhelmed by the high acidity caused by the failure of the ALD to add any alkalinity to the influent.

The Centerville "C" wetland exhibited the highest treatment efficiency of the three wetlands. According to the Schafer and Associates (1994) report, the wetland effluent showed improvement in water quality from influent to effluent in the reduction of iron (58 mg/L to 5.0 mg/L), aluminum (263 to 18.7 mg/L), sulfates (3150 to 2150 mg/L), but an increase in zinc (2.84 to 5.0 mg/L). However, as in the French Coulee wetland, the manganese increased from 2.54 to 4.55 mg/L. The effluent showed improvement in total dissolved solids and an increase in pH from 2.7 to 4.4. Currently, water quality leaving the wetland is improving. This indicates that the wetland is performing but not at a level adequate to meet livestock drinking standards. The wetland effluent is discharged into a stock tank for cattle.

The inadequate ameoration of the acid load in the mine drainage was to severe for the passive wetland to treat effectively. The wetland may have been to small to function without influent pretreatment or additional alkalinity added to the water before it enters the wetland. By raising the pH of the drainage and by adding alkalinity, the wetland may

be able to bring the effluent discharge into compliance. The substrate bacteria efficiency would be greatly enhanced if the pH of the influent was raised to pH 5 and the metals will be more readily complexed at pH 4, aiding in their precipitation. Without pretreatment it is uncertain how long the wetland will continue to provide even the present level of treatment.

Seeding vegetation on the site was completed in July, 1992. The grass species are reestablishing. Cattails and native plants were established in the cells and should be monitored over the next few of years. A field visit in the winter of 1995, showed successful cattail establishment in Cells 1 and Cell 3 but Cell 2 cattail development was marginal. No explanation can be offered for the poor cattail establishment in Cell 2.

During a field visit, samples of the substrate were collected from Cell 1 and Cell 2. Carbon and nitrogen concentrations were determined. The results were compared to a sample taken from the Giffen wetland, a natural wetland treating AMD. The test showed that the ratio of carbon to nitrogen (C:N) was not out of balance for Cell 1 or for Cell 2. The Centerville "C" wetland carbon to nitrogen ratio is comparable to the Giffen natural wetland (Table 3). A high C:N ratio does not seem to be the factor limiting cattail establishment in Cell 2.

Table 3. C:N Ratio comparison of Centerville cells to a natural wetland.

Wetland	Carbon %	Nitrogen %	C-N Ratio
Centerville Cell 1	37.92	1.41	26.9 to 1.0
Centerville Cell 2	14.35	1.08	13.3 to 1.0
Giffen Natural	4.43	0.31	14.3 to 1.0

Stockett SE Site

Stockett Northwest Interceptor System

Four main interceptor drain lines moved AMD to the northeast service hole and the north wetland channel (Figure 3). Interceptor drains were located in several places. One was placed in the southeast coulee, another on the east facing hill, a third on the upper slope of the NW facing hill, and a fourth drain on the lower slope of the NW facing hill (Figure 3).

Acid mine drainage sampled during construction showed higher dissolved oxygen concentrations than expected in the east hill interceptor and in the lower slope NW facing hill interceptor. These two water sources revealing high dissolved oxygen were joined and bypassed around the ALD in the NE service hole. The two lower dissolved oxygen water sources were joined and were designed so that they may flow

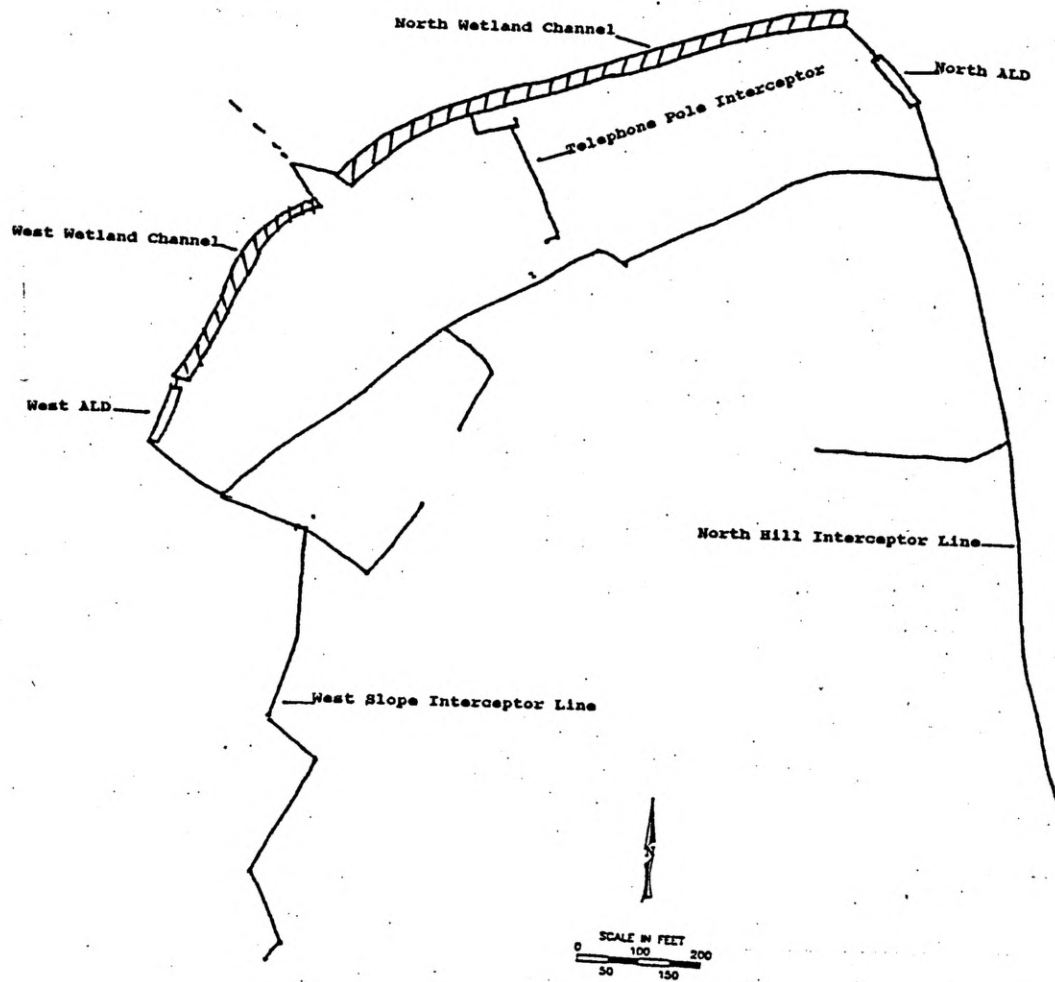


Figure 3. Planview of Stockett AMD treatment system
(Adapted from Schafer and Associates 1994).

to the ALD or they may bypass the drain at the NE service hole. At a junction point below the NE service hole, the sources of bypass water and effluent from the ALD were joined to flow into the wetland channel. A third source of water was intercepted low on the west slope of the Stockett site by a structure called the Telephone Pole Interceptor. The water from the Telephone Pole Interceptor entered directly into the north wetland channel.

Stockett SW Interceptor System

The southwest interceptor system collected water from three main interceptor drains and flowed into Service Hole #1. From Service Hole #1, the drainage water can bypass the ALD to outlet at the west wetland channel, flow through the ALD and to the outlet in the wetland channel, bypass both the limestone trench and west wetland channel and enter the road side ditch.

The Stockett wetland channel was designed to test the concept of a long, linear wetland for future AMRB projects. Conceptually, the design was for a continuing series of oxidizing and reducing conditions to treat the acidic water. To achieve alternating oxidizing and reducing conditions baffled check dams were installed along the channel grade at 50 foot increments. The idea was to maximize effective treatment area because of the severely limiting topography.

The design used the existing topographic features and a surface exposed liner rather than installing a buried liner which would use valuable space with back fill material to cover an non-exposable liner.

After construction began, AMD seeps exhibited higher than expected levels of dissolved oxygen. The higher dissolved oxygen levels in the seeps caused concern for the longevity of the original ALD. The longevity of the ALD would be reduced by increased metal precipitation (Peccia and Associates 1993). The ALD's were built to original specifications of the bid hoping that the high dissolved oxygen drainage would equilibrate to a lower level of oxidation (Peccia and Associates 1993).

The Stockett SE wetland was completed in July, 1992. The wetland consisted of two linear plastic lined drainage channels which discharge at a common point. The original plans called for the construction of a wetland cell below the drainage channels to be added later to the system. The total AMD flow was 5 gpm and the total effective treatment area of the channels was less than 0.1 acre. Each channel is fed by interceptor drains and the collected AMD stream was to be pretreated in an ALD. Before the construction was completed the system had problems with the interceptor drains. The drains were blocked by precipitates and a four inch perforated pipe had collapsed. Problems were blamed on improper installation. They were corrected and AMD

treatment started in July, 1992. The North ALD showed no pretreatment and the West ALD system resulted in increased fouling. The ALD system was bypassed during the winter of 1992 and flow resumed in the spring of 1993 but the ALD was by passed in the summer due to flow restriction and decreased chemical performance. In October 1993, water was observed flowing at the surface especially on the north side.

A problem also existed in the North wetland channel collection system. Significant slope failure, erosion and a rechanneling of discharge indicated a failure in the system. Effective pretreatment was never observed in the North Limestone trench or ALD. The problem may have been in the design and construction, as analytical data indicated that all the flow was bypassing the ALD and going into the "high oxygen" header.

Water chemistry in the West limestone trench (ALD) indicated that aluminum was being removed as a precipitate, ferric iron was being precipitated, and sulfate was being removed (as CaSO_4). West Trench was no longer providing enough alkalinity to provide total removal of iron. At the time of this study, the removal of iron was the only significant change in chemistry from influent to effluent (1170 mg/L to 878 mg/L). All other chemical parameters showed little to no improvement (Schafer and Accociates 1994).

WETLAND PROCESSES

Aerobic Wetlands Processes

It was previously thought that the removal mechanism for metals in wetlands was from humic material adsorption or through metal uptake by plants. At present, it is thought that the dominant removal processes in wetlands are aerobic and anaerobic microbial activity (Wildeman et al. 1993). Both processes can be used in a constructed wetland system to treat mine drainage. Aerobic wetlands have been used in the eastern United States to treat coal mine drainage (Brodie et al. 1992). Wetlands are being examined in the west as a treatment for coal as well as metal mine drainage. A comparison of wetland types in Table 4 shows the emphasis of each process (Wildeman et al. 1993).

Aerobic systems perform best with drainage at pH 5.5 and higher and with iron and manganese metals (Wildeman et al. 1993). Anaerobic systems can treat mine drainage at pH 2.5 and are capable of removal of other heavy metals such as copper, zinc, cadmium, and lead. The ability of the anaerobic processes to perform at low pH's and removal of a broader range of metals commonly found in metal mine

drainage makes anaerobic systems more appropriate processes for western mine drainage problems.

Table 4. A comparison of aerobic and anaerobic wetlands.

WETLAND COMPARISON	
Aerobic	Anaerobic
Emphasize oxidation	Emphasize reduction
Surface flow of water	Subsurface flow of water
Oxide precipitates	Sulfide precipitates
Process lowers pH	Process raises pH
Operates best at pH > 5.5	Can be effective at pH 2.5
Might freeze in winter	Can operate in winter
Removes Iron quite well	Removes heavy metals

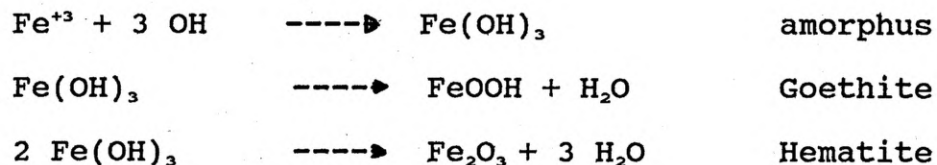
The aerobic systems emphasize oxidation processes which can cause metal oxides to precipitate. Typical microbial mediated reactions in the aerobic zone include (Wildeman et al. 1993):



Aerobic processes produce hydrogen ions and can lower the pH unless there is sufficient alkalinity in the system to buffer the solution.

The oxidation process is followed by the precipitation of hydroxides, generally, iron and aluminum compounds. The

oxidized form of iron, ferric iron, produces the following precipitates (Wildeman et al. 1993):



The solid precipitates can be gelatinous and have a strong capability for adsorbing other metals onto their surfaces. Amorphus substances have polymeric properties and a gelatinous appearance. The process of metal adsorption can be used in polishing and buffering stages in passive wetland treatment.

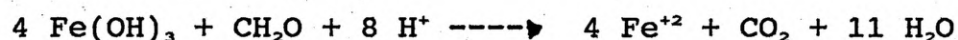
Bacteria can play a role in the aerobic zones by efficient removal of iron and manganese by hydroxide precipitation under oxidizing conditions. This removal is most readily achieved by microbial catalysis in the aerobic zone. Many bacteria oxidize iron but the most important ones are in the Ferroxidans family. Ferroxidans can tolerate pH's less than three and exist on inorganic nutrients of ferrous iron, carbon dioxide and ammonia. Other bacteria species are capable of oxidizing ferrous iron when the pH reaches 5 but they require organic material for nutrients (Wildeman et al. 1993).

The Tennessee Valley Authority (TVA) is operating surface wetland systems for treating acid mine drainage (Wildeman et al. 1993). Microbial catalyzed oxidation reactions are thought to be the primary processes for iron

removal. Five of the operating wetlands have produced effluent meeting discharge limits. However, these wetlands have influent of levels iron from 11 to 69 mg/L, manganese 5-14 mg/L, and pH 5.5 or above (Wildeman et al. 1993). The AMD in the western United States have higher metal levels and lower pH values than the AMD being treated in successful oxidizing TVA wetlands.

Anaerobic Wetland Processes

Anaerobic wetlands use the chemical process of microbial bacteria and adsorption on organic substrates to treat acid mine drainage in these systems. Bacteria in the substrates are thought to be the source of the major processes treating acid mine drainage. Bacteria in the substrates, under reduced conditions, consume sulfates and hydrogen ions, thereby raising the pH and producing sulfides. The sulfides are precipitated. Anaerobic wetlands can treat highly acidic mine drainage (pH at 2.5) and remove other metals as well as iron. Typically anaerobically mediated reactions include :



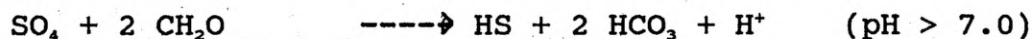
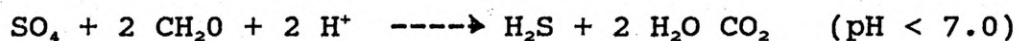
The flow in anaerobic wetlands should be in a downward mode so that acid mine drainage can be treated in the substrate. The process does not depend on vegetation except for the addition of organic matter to the substrate as a carbon source for bacteria. Almost any source of carbon may be used for this purpose.

Metal uptake by plants, and specifically *typha* species, showed little impact on total metal loads (Wildeman et al. 1993). However, the literature reveals conflicting opinions about the importance of vegetation in metal removal. Anaerobic bacterial activity can cause metals to precipitate as sulfides or carbonates. A byproduct of bacterial activity is bicarbonate alkalinity which would raise the pH of acidic water and stimulate hydrolysis reactions (Hedin 1991).

Composted wetlands have increased the pH of AMD with a simultaneous decrease in the concentration of sulfates however, sulfate removal can be variable (Hedin 1991). Changes in sulfate removal has shown seasonal variation. Reasons for this variability include low winter temperatures effecting bacteria activity, changes in the solubility of oxygen in the aerated water, and plant growth decreases. Variability in sulfate reduction as experienced at other mines may also be attributed to higher drainage flows during winter and spring (Wildeman et al. 1993). Alkalinity is generated by sulfate reduction processes. The process can

generate up to 200 mg/L of sulfides available for metal complexation (Skousen et al. 1992).

The bacteria most capable of catalyzing the chemical reaction for sulfate reduction are in the *Desulfovibrio* family. These bacteria need the correct range of hydrogen ion concentration and proper reduced/oxidized environment to grow and produce. Sulfate reducing bacteria can tolerate a temperature of -5 to +50 degrees centigrade making them effective in cold climates. Organic nutrients such as lactic acid or cellulose material are needed by bacteria for growth. The activity of these sulfate-reducing bacteria is severely repressed below pH 5 but they can tolerate a pH range of 5 to 9.5. It is important to establish their populations before adding AMD to the substrate. If AMD is added to the substrate before the bacteria have had sufficient time to establish, the bacteria may not be present in sufficient numbers to treat the AMD. Relatively clean water should to be added to the wetland before the addition of AMD. Sulfate reactions include:



Bacteria can create and maintain their own microenvironments; when pH is low H_2S and CO_2 redissolve into solution limiting the pH decrease and when the pH is too high $\text{H}_2\text{S}/\text{H}_2\text{CO}_3$ neutralizes the basic conditions. The reactions generate hydrogen sulfide which reacts with

copper, lead, antimony, cadmium, and iron creating sulfates.

A population count of sulfate reducing bacteria can indicate whether the proper environment is being maintained on a long term basis and if sulfate reduction is occurring. In addition to population counts, it is also important to determine the species of bacteria and in what ratios they occur. This provides a good indication of the maintenance of the correct pH and Eh. Removal of heavy metals requires that sulfide, the product of sulfate reduction, come in contact with the mine drainage. Therefore, the flow rate through the substrate should be relatively constant (Wildeman et al. 1993). The flow should not be so great as to overwhelm the system or so low as to threaten the maintenance of the reactions. Evidence of sulfate reduction includes a decrease in sulfate concentration in the effluent and a decrease in Eh in the effluent compared to influent. Sulfate reducing bacteria require reducing conditions.

Sulfate reduction and sulfide retention processes are present in anaerobic wetlands. The formation of highly insoluble reduced precipitates removes metals, increases pH and reduces sulfate concentration. The success of the processes is dependent upon drainage flow through the anaerobic portion of the wetland substrate and the continuous flooding of the wetland. These two actions are necessary to prevent the oxygenation of the products of reduction (Wildeman et al. 1993).

Sulfate reduction and metal oxidation are mutually exclusive processes. Wetlands need to be staged, since sulfate reduction raises pH, this process should occur before the oxidizing process (Wildeman et al 1993). Anaerobic systems remove manganese ion as $MnCO_3$. In a constructed wetland, manganese ion removal needs to be 2.5 to 3.5 times the area necessary for removal of a comparable quantity of iron per milligram (Wildeman et al. 1993).

At pH values from 4 to 6, depending on the type of ion, efficient complexation of metal ions occurs. Adsorption of a metal ion also implies desorption of another ion. There is a limit to the amount of metal adsorption that can take place. Organic exchange sites could serve as temporary retention of metal cations on the substrate. This can provide an increased residence time for microbial mediated metal removal process. Stable species of sulfur allow sulfate bacteria to operate. Therefore, the proper conditions of oxidation/reduction (Eh) and hydrogen ion concentrations (pH) must be maintained for effective treatment.

Acid Mine Drainage Chemistry

The study wetlands have not adequately treated AMD. One possible explanation for this problem is the extreme chemical nature of the drainage entering the system.

Constructed wetland technology was first developed in the eastern United States. Tables 1 and Table 5 display chemical properties of AMD at the study sites and from the eastern United States. Generally, the influent of the eastern AMD sites had higher pH values and lower iron and aluminum levels in comparison to the western study wetlands. Sulfate concentrations at the study wetlands ranged from two to ten times as high as the sulfate levels in eastern wetland sites. In example, concentrations at the Stockett Site were almost eight times as high as the eastern AMD sites.

The realization that the amount of acidity and metals in the AMD of the three wetlands is much greater than the amount normally encountered at other locations in the United States can help us understand their lack of performance. Centerville "C" AMD influent compares favorably with sites where AMD has been successfully treated while the AMD at Stockett is several orders of magnitude more chemically enriched and may be more difficult to treat.

Sizing of Study Wetlands

The science of sizing a wetland for AMD treatment is in the developmental stage. To determine the size of a wetland for AMD treatment, the amount of acidity and the concentration of metals in the drainage must be determined.

Table 5. Average chemistry of influent and effluent water of various wetlands in eastern United States (Hedin 1991).

Site	Substrate	Flow, L/min	Average influent			Average effluent		
			pH	Fe ⁺	Mn ⁺	pH	Fe ⁺	Mn ⁺
Cedar	none	568	6.3	95	2	7.0	45	2
Cottage	hay bale	180	3.5	20	4	4.0	6	4
Donegal	compost	412	7.7	5	8	8.0	<1	1
Emlenton	manure	61	4.5	96	85	3.0	20	95
FH	compost	15	2.6	200	10	3.0		
Keystone	none	7900	6.5	37	1	7.0	30	1
Latrobe	compost	86	3.5	125	32		50	
Piney	none	388	6.0	<1	16	6.0	<1	10
Shade	limestone	21	6.5	<1	34	7.0	<1	5
Simco	compost	328	6.5	111	2	6.0	42	2
Somerset:								
Summer	compost	27	3.7	202	59	6.0	5	?
Winter	compost					4.0	100	59
TVA:								
950-1&2	none	83	5.7	120	8	6.5	1	2
IMP 1	none	73	6.1	69	9	6.7	1	2
IMP 4	none	131	4.5	110	25	4.6	3	4
Wif 018	none	70	5.6	150	7	3.9	6	6
Wif 019	none	492	5.6	18	7	4.3	3	6
IMP 2	none	1016	3.1	40	13	3.1	3	14
IMP 3	none	250	6.3	16	5	7.0	<1	1
RT-2	none	277	5.7	45	13	6.8	1	2
950-NE	none	385	6.0	11	9	6.9	1	1
Kif-6	none	379	4.2	170	4	2.9	83	5
Cof-013	none	288	5.7	1	5	6.7	1	14

+ Fe and Mn concentrations-mg/L

More progress has been made sizing aerobic wetlands than anaerobic wetlands (Hedin 1991). Several different methods are used to size either type of wetland. In northern Appalachia wetlands have been sized hydraulically using the standard of five square meters of wetland for each liter per minute of flow. This standard is applicable for drainage with flows of less than 40 Lpm, pH values less than 4, and iron concentrations less than 50 mg/L. The formula is based on hydraulic loading and chemical loading. None of the three wetlands in this study meet all of these criteria. The study sites contain 300 mg/L to 1100 mg/L more iron than the recommend iron concentration. Therefore the previous method of sizing wetlands is not recommended for this study area.

Tennessee Valley Authority has built numerous wetlands and utilize a formula based on iron loading (Brodie 1990).

$$\text{Aerobic wetland size} = \frac{[\text{Fe conc. (mg/L)} \times \text{Flow (L/min)} \times 1.44]}{\text{Recommend Chemical Loading Rate}}$$

The units of the recommended chemical loading rate is grams of contaminant removed per day per meter (gdm). Brodie and others have found that the gdm can have a range of two to ten gdm. Brodie suggested that this formula is used for AMD of pH > 4.5. If the wetland is to treat manganese the size should be increased by five or six times. The formula takes into account iron loading and not other metals or acidity.

Hedin proposed an adjusted area loading formula:

$$\text{Fe gdm} = \frac{1.44 \times \text{Flow (L/min)} [\text{Inflow Fe conc} - \text{Outflow Fe conc}]}{\text{area (m}^2\text{)}}$$

The formula takes into account only iron loading, other researchers suggest substituting total acidity (sum of Fe, Al, and H⁺). Using the formula based on iron loading and the conservative 2 gdm, the French Coulee should be at least 35.1 hectares (95 acres) in size and Centerville "C" should be 1.44 hectares (3.9 acres). These estimates are conservative and are based on total treatment by aerobic processes. If the wetlands are to be used in an aerobic treatment flow the size of the wetlands are too small.

This discussion has focused on sizing for aerobic processes and has not taken into account the anaerobic process. The anaerobic process relies on microbial sulfate reduction. A sulfide loading factor of 300 nanomoles/cubic cm/day for the amount of sulfide generated and concentration of heavy metals in the drainage is used to adjust the flow and to predict size of anaerobic substrates (Wildeman et al. 1993). This factor should be adjusted so that sulfide is always in excess. Using the volumetric loading method, the estimated size of the French Coulee site should be approximately 17400 square meters or 24 to 32 times larger than its present size. The Centerville "C" site should be approximately 14 times (or 5980 square meters) larger than

its present size if it is to completely treat AMD anaerobically. Of course, treatment of AMD is a combination of aerobic and anaerobic processes so these estimations are inflated. However, the conclusion that can be drawn is that the wetlands were built too small to treat AMD adequately without additional treatment.

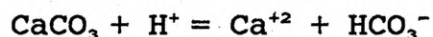
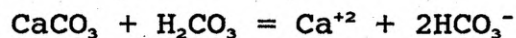
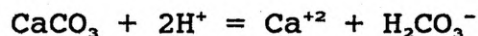
POTENTIAL TREATMENT OPTIONS TO ENHANCE WETLAND PERFORMANCE

Anoxic Limestone Drains

To aid in successful treatment of acid mine drainage alkalinity can be added to passive wetland systems. One method of generating alkalinity in mine drainage is by anoxic limestone drains (ALD). Anoxic means without oxygen. An anoxic limestone drain is a shallow, oxygen starved limestone filled trench which is sealed from the atmosphere. In the reduced environment, limestone will not become armored by oxidizing iron and the limestone will continue to add alkalinity to AMD. The concept of ALDs was discovered by the Tennessee Valley Authority when a 7.6 meters (25 foot) high earth dam was built over an existing coal haul road constructed of crushed limestone (Brodie et al.1992). Tennessee Valley Authority decided to investigate and found that the AMD was being treated by raising the pH and adding alkalinity. The dissolution of limestone from the road was observed in the rounded edges of the limestone but the limestone showed no evidence of coatings or armoring.

Anoxic limestone drains pre-condition the drainage water by adding alkalinity which raises pH and buffers the water. Limestone dissolution neutralizes hydrogen and aluminum ions, two components of acidity, thereby decreasing

the amount of acidity in the drainage. The following chemical reactions occur in an ALD (Brodie et al. 1992):



In the drain, the presence of bicarbonate (HCO_3^-), from dissolution of limestone mixes with aluminum and hydrogen ions. Hydrogen ions react with HCO_3^- to form H_2O and CO_2 . Aluminum ions reacts with HCO_3^- to form $\text{Al}(\text{OH})_3$ and three CO_2 (Skousen et al. 1992). These reactions consume some of the generated alkalinity and the aluminum compounds precipitate in the drain. The precipitation may clog the drain but until then the precipitation keeps aluminum away from fish and plants both of which are highly susceptible to the toxicity of aluminum.

The chemistry of the AMD must meet certain parameters before an ALD can be utilized to treat it. To allow the limestone to dissolve without becoming armored, reduced conditions must be maintained in water coming into the system. The water entering the system must be in an anoxic condition. There are several ways to measure the anoxic conditions: 1) dissolved oxygen in the water, 2) ratio of ferric iron to ferrous iron content, and 3) Eh. The dissolved oxygen content should be less than 2 mg/L. For an ALD to work effectively, iron should be in the ferrous state. In this state iron will not precipitate at the

higher pH levels generated by the ALD. Ferric iron in the acid water will precipitate ferric hydroxides in the drain when alkalinity is generated, coating and plugging the drain. Aluminum compounds will precipitate in the water at or above pH 5 through hydrolysis and do not require oxidation. Therefore, aluminum concentrations should be at or below 25 mg/L to prevent clogging. It has been suggested that the aluminum and the ferric iron content together should be 25 mg/L or less. However, some aluminum precipitates have been observed leaving the ALD's as gelatinous flocs and the flocs don't seem to have an affinity for limestone. The Eh measurement of the AMD should be at zero or less (Skousen 1991). If these chemical conditions have been met the acid mine drainage may be discharged into an ALD for treatment.

Variability in efficiency in alkalinity generation and metal retention exist between ALD's (Hedin et al. 1994). One drain can add up to 300 mg/L of alkalinity and raise the pH over six but these are under optimal conditions (Hedin 1991). The acidity in ALD sites in West Virginia decreased by 50 to 80 (Skousen et al 1992). If constructed and maintained correctly the ALD longevity should be up to twenty years or more (Skousen 1991).

Discharged AMD from an ALD is not completely treated and cannot be immediately discharged into receiving streams. The ALD effluent still contains metal compounds to be

removed by precipitation. During the precipitation process acidity increases because a hydrogen ion will be produced when ferrous iron and manganese oxidize, hydrolyze and precipitate as hydroxides. The oxidizing process must be buffered by additional alkalinity added by the ALD or from another source. Anoxic limestone drains are not stand alone systems and discharges should be released into a ditch or a settling pond. The discharge can be directed into a passive wetland for removal of metals by precipitation. The alkalinity added by the ALD enhances the performance of passive wetland.

Anoxic limestone drains are shallow limestone filled trenches. Constructed trenches have ranged from 0.6 to 1.5 meters (2 to 5 feet) deep, 0.6 to 2.7 meters (2 to 9 feet) wide, and range from 30.3 to 606.1 meters (100 to 2000 feet) in length (Skousen 1991). The trench may be covered with geotextile sheet to keep oxygen out of the drain. The fabric should be strong enough to prevent puncture from the construction over the top of the trench. Oxygen can be scavenged from the trench by placing hay bales in it. Enclosing the hay bales in plastic separates the hay and ALD to prevent clogging of the limestone by hay fragments. The decomposition of hay enhances the reduced environment in the ALD. The added organic material will decompose using up oxygen and producing carbon dioxide. The trench should be covered with an amount of soil (usually 1 meter) to

guarantee oxygen does not enter the trench. A clay is recommended for the cover soil. The cover should be crowned to allow for subsidence in the drain as limestone dissolve. The site should be revegetated and waterbars used to prevent erosion of the cover.

Anoxic limestone drains have been successfully tested in the eastern United States as a pretreatment for acid mine drainage prior to discharge into passive wetland systems. Similar anoxic limestone drains were constructed at the Centerville and Stockett sites. They failed almost immediately due to clogging. The ALD failures were probably caused by AMD entering the treatment systems that had low pHs and little alkalinity. Clogging could have been due to the large concentration of ferric iron and large concentrations of aluminum.

An alternative to using an ALD to add alkalinity to AMD is to use a passive wetland. The passive wetland could bring the drainage into a reduced state by removing dissolved oxygen. An anaerobic wetland would scavenge oxygen from the water and precipitate ferric iron. The anaerobic wetland effluent can be introduced into an ALD through closed pipes. A passive system of post ALD treatment had been operating effectively for over a year (Skousen et al 1992).

French Coulee and Centerville ALD Implementation

The anaerobic substrate from the Centerville and French

Coulee wetlands could be used to pretreat the mine drainage before being put into an ALD. The effluent from these sites showed some improvement in the reduction of metals, however, there is little improvement in pH. Further investigation of the effluent from the anaerobic substrate is needed to determine if the effluent could meet the chemical parameters necessary for successful treatment in an ALD. Dissolved oxygen, ferric iron content, aluminum concentration, and Eh values need to be measured in the water in the anaerobic substrate. If the effluent meets the conditions for treatment in an ALD, the effluent from the substrate of the wetland must be conducted from the wetland to the ALD without exposure to oxygen. If this treatment method is pursued, an oxidation basin will be necessary to hold the AMD before it is discharged into the natural system.

If building of a post wetland ALD is a viable treatment option additional testing was required. To test the reduced conditions of the substrate the ratio of ferric to ferrous iron, pH, Eh, and dissolved oxygen must be determined. During the spring of 1995 the author attempted to test these substrate effluent parameters at French Coulee and Centerville. Using PVC well screen pipe and a suction device with nitrogen gas to displace oxygen, water was collected from the reduced zone of the wetland substrate. Care was taken not to contaminate the sample with oxygen. The French Coulee and Centerville "C" Cell 3 substrates were

tested. The results are shown in Table 6.

Table 6. Comparison of chemical parameters of the oxidized and reduced areas of the French Coulee and Centerville substrates. Field data-1995.

	pH	Eh mv	Fe ⁺² mg/l	Fe ⁺³ mg/l	Total Iron mg/l	C ^{o1}	DO ² mg/l
French Coulee Oxidize	2.4	236	230	9.9	240	20.0	7.2
French Coulee Reduced	4.4	137	160	1120	1280	11.8	5.0
Centerville Oxidized	3.6	244	97.4	2.6	100	22.4	8.0
Centerville Reduced	6.8	39	9.61 7.7 7.5	54.4	64		

1. Temperature
2. Dissolved Oxygen

These chemical parameters indicate that the anaerobic zone was reducing but the conditions were not sufficiently reducing the AMD to allow an ALD to treat the AMD. The dissolved oxygen content for the wetlands is too high to be treated in an ALD. The chemical properties of Eh and ratio of ferrous iron to ferric iron are not within parameters for successful treatment.

Stockett System

The treatment system at Stockett did not have any substrate to scavenge oxygen from the drainage. Since there

was no wetland substrate, Stockett did not lend itself to a post ALD system. An ALD for pretreatment at Stockett has already been tried and has failed.

Open Limestone Channels

When the water quality of mine drainage precludes the construction of an ALD, open limestone channels (OLC) maybe a simple alternative to the use of ALD's. Open limestone channels are open, free flowing conduits lined with coarse limestone (< 4 inch diameter). The original purpose of the construction of these OLC was for water conveyance and erosion control, not for AMD remediation. The treatment of AMD was an unexpected benefit of them.

Researchers looked at five OLC sites in West Virginia (Ziemkiewicz et al. 1994). All sites showed acidity reduction. Some sites only partially treated AMD while others achieved 100% treatment of the AMD. At one study site, noted as the Webster AML site, the drainage from three portals had a pH of 3, with acidity measuring 100 to 300 mg/L. After passing through the OLC, the calcium in the drainage was increased by 450%. Iron, manganese, and aluminum were entirely removed. After two years of field study, the fully armored limestone was 1/5 as soluble as unarmored limestone but armored limestone was still able to provide remediation.

A critical factor in the effectiveness of the OLC is the length of the drain. Knowledge of the minimum level of acceptable cleanup, identified limestone dissolution rate, an AMD situation allows for the estimation of the length of the channel and the tons of limestone needed for remediation.

Open limestone channels could be an inexpensive pretreatment step. A channel should be constructed to withstand high flow periods and should be nearly maintenance free. Open limestone channel should be constructed on steeper hillsides or in steep dry stream channels. The higher velocities will carry the floc-precipitation of ferric and aluminum hydroxides downstream minimizing armoring and sludge deposition on the limestone. Low lying stretches of low current velocity might cause the drain to be buried under precipitating floc and sediment, isolating it from the AMD.

The construction criteria will be the same used in Table 7. To use the calculation method (Table 7) from Ziemkiewicz (1994) flow rates and acidity must be known for all AMD to be treated. The acidity of the AMD must be 1000 mg/L or less. The stream channel cross section has the dimensions of 0.9 meters (3 feet) deep by 3 meters (10 feet) wide. Assuming the limestone will dissolve at 20% maximum efficiency when armored (Ziemkiewicz et al. 1994) a one hour of residence time will produce 90% efficiency of treatment

Table 7. Construction criteria for OLC (Ziemkiewicz et al. 1994).

Construction criteria of open limestone channels for treating AMD*. The calculations are for a stream cross-section of 3 ft deep by 10 ft wide and an acidity concentration of 1000 mg/l, and assumes that limestone will dissolve at 20% maximum efficiency when armored. Under the given conditions, the data estimate the length of a channel and the tons of limestone which will provide 90% neutralization of the acidity over a 1 hr residence time or 100% neutralization after 3 hours residence time.

Conditions Flow	Velocity ft/sec	1 or 3 hrs Contact Time		1 or 3 hrs Contact Time			
		Channel Length (ft)		Tons of Limestone			
		1 hr	3 hr	100% Eff	20% Eff	1 hr	3 hr
gpm		-----ft-----		1 hr	3 hr	1 hr	3 hr
100	0.02	67	201	169	508	847	2542
200	0.04	134	401	339	1017	1695	5085
300	0.06	201	602	508	1525	2542	7627
400	0.07	267	802	678	2034	3390	10170
500	0.09	334	1003	847	2542	4237	12712
600	0.11	401	1203	1017	3051	5085	15254
700	0.13	468	1404	1186	3559	5935	17797
800	0.15	535	1604	1356	4068	6780	20339
900	0.17	602	1805	1525	4576	7627	22882
1000	0.19	669	2006	1695	5085	8475	25424

*From Pearson and McDonnell, 1974. 0.3 meters-1.0 feet

and a three hour residence time will produce 100% efficiency (Ziemkiewicz et al. 1994).

Flow rates for each wetland were calculated using data from the United States Geological Survey and from Schafer & Associates 1994. The flow rates are necessary to determine the length of OLC for each wetland in the study area. The flow rate parameter indicated an OLC length of 61 meters (201 feet) and 2542 tons of limestone should treat AMD at all three sites. The acidity of the AMD at the three sites is greater than the 1000 mg/L level used by the West Virginia researchers (Table 7). French Coulee and Centerville AMD acidity levels are 1 1/2 to 2 times the indicated threshold. The Stockett SE AMD is 10 times the preferred level. Research is needed to determine if increasing the length of channels or time of treatment will ameliorate the extra acidity. Additional research is needed to determine if the high acid levels are a problem for using OLC to treat the AMD at the study sites.

The use of OLCs may be an alternative that will improve treatment of AMD at the French Coulee, Centerville and Stockett wetlands. Open limestone channels could be used to pretreat mine drainage before being discharged into the Centerville and French Coulee wetlands or as a post treatment of effluent from the wetlands. At the Stockett site, the North and West Channel of the Stockett system could be lined with limestone to treat the AMD. The channel

would have to be at least 61 meters (201 feet) in length. The channel should be steeply constructed to help reduce sedimentation. The OLC could be visually impacting to residents of Stockett and to highway users. Obtaining permission from the land owner may also be an obstacle to construction.

The OLC should increase the pH and add needed alkalinity to the influent entering the wetland which would enhance wetland performance. The wetland could then be used to "polish" the drainage by allowing precipitation to be filtered or removed before discharge in to the natural system. If the discharge from the OLC was clean enough then the wetland could be bypassed.

French Coulee Wetland

Another option for treatment would be to use the OLC as post treatment system. The OLC would be constructed in an existing ditch on the east side of the railroad tracks bordering the French Coulee wetland. At present the ditch conveys AMD from the mine portal to Belt Creek. The public would be more likely to accept this option since the ditch is already present. Further research is needed to determine if the OLC can be used as a stand alone system or if a settling pond will be needed before discharge into the creek. Like the pretreatment OLC, the post-treatment OLC must be between 61 to 121 meters (201 and 400 feet) in

length to maintain adequate treatment for the AMD after the channel becomes armored.

Centerville Wetland

The Centerville wetland site also has the potential to use an OLC as a pretreatment or as a post-treatment system. The hillside below the surface collection pond provides enough length and slope, for the pretreatment of AMD. To treat the AMD, the length of the channel needs to be between 61 to 91 meters (201-300 feet) in length to treat the large amount of acidity in this drainage. The wetlands could then be used as a settling pond for precipitates before discharge into the local watershed. The small drainage ditch below the stock tank and the creek would be a good location for a post-treatment OLC. However, a "polishing" pond would need to be constructed before discharging effluent into the creek.

Stockett Wetland

The Stockett SE site has two lined channels already constructed at the site. The channels could be converted to OLC's. The North Channel is relatively level which could become obstructed with sediment or floc precipitation. The North Channel is approximately 113.6 meters (375 feet) in length and the West channel is approximately 242.4 meters (800 feet) in length. The AMD at this site contains a large

amount of acidity approximately 10 times greater than the 1000 mg/L used to calculate desirable length of an OLC. If 61 meters (201 feet) treats 1000 mg/L then it may be assumed that 10 times the amount of acidity may be treated with 10 times the length of open limestone channel or 91 meters (2000 feet). If 91 meters (2000 feet) of channel is needed to treat the acidity, cost and area limitation could be a concern at this site. However, the flow rate at the site suggests that the site can be treated with 61 meter (201 feet) of OLC.

Sustained Alkalinity Generating System

Debate and research continues on the effectiveness of the passive wetland treatment systems, as some systems function while others don't. After reviewing the literature it becomes clear that the addition of alkalinity may be essential to the success of the passive wetland treatment systems. The addition of alkalinity becomes increasingly important as acid mine drainage chemistry intensifies. The development of successive alkalinity production systems (SAPS) can reduce the currently accepted area treatment needs of a passive treatment system design.

Kepler and McCleary (1994) developed sustaining alkalinity generating systems by combining the designs of an ALD and anaerobic wetland systems. A layer of limestone is

overlain by a layer of organic substrate in a SAPS design. The acid mine drainage is ponded over the top of the substrate. Water pushing down through the organic substrate becomes reduced and flows through the limestone layer picking up alkalinity through limestone dissolution (Figure 4). The water is in an anoxic state when it flows through the limestone so the limestone does not become armored. A drainage pipe is placed in the limestone layer to discharge the effluent into the next cell of a wetland or into an oxidation pond to precipitate metals. Pressure from the downward force of standing water above the substrate should reduce the risk of physical plugging as compared to waters flowing laterally through conventional ALDs (Kepler and McCleary 1995). Aerobic wetlands should be utilized between SAPS components to aid in removal of suspended solids (Kepler and McCleary 1994).

Kepler and McCleary (1995) studied four SAPS systems operating in Jefferson County, Pennsylvania. All sites showed an improvement in pH and an increase in alkalinity. Some sites showed improvement from pH 3.5 to 6.3 and removal of up to 275 mg/L of acidity. The effluent contained low values of dissolved oxygen, low values of Eh, and the system discharged iron in the ferrous state. The discharge of iron in the ferrous state prevented the armoring of limestone from becoming a problem and the discharge pipes of

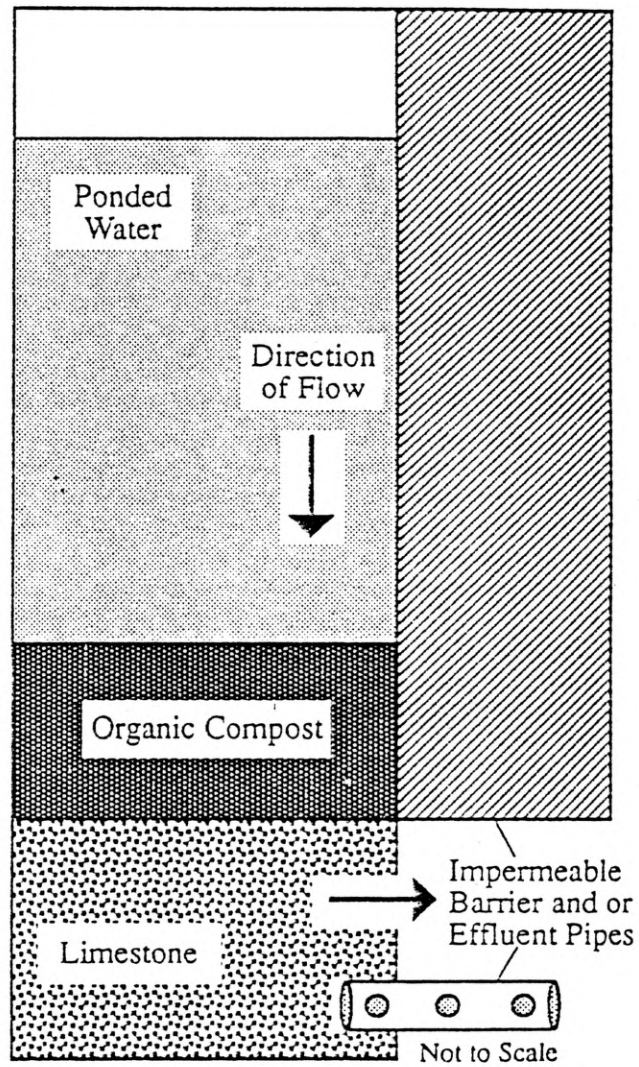


Figure 4. Sustained alkalinity producing design (Kepler and McCleary 1994).

the system never accumulated orange ferric oxihydroxide precipitates.

French Coulee Wetlands

At the French Coulee wetland, Cell 1 could be converted to a sustaining alkalinity generating system. At present cell one has been reduced in its capacity to treat acid mine drainage and has very little vegetation. The cell could be drained and the cell substrate removed. The cell could then be replaced with limestone overlain by organic substrate. The cell would need effluent discharge pipes installed in the limestone layer to direct effluent to the next wetland cell. The remaining two cells could be used to treat the acid mine drainage and act to "polish" the AMD before discharge into the natural system. The addition of alkalinity should produce a more favorable influent to be treated by sulfate bacterial activity and to produce enough alkalinity to counter act the lower pH's produced by the oxidizing metal precipitation. The optimum residence time in the SAPS should be at least twelve hours.

Centerville Wetlands

French Coulee and Centerville wetland performance could be enhanced by modifying a wetland cell to a sustaining alkalinity producing system. The Centerville "C" Cell 2 could be used to generate alkalinity into the drainage. At present, the vegetation in Cell 2 has established poorly.

This cell could be converted to a SAPS system. The cell substrate could be removed and replaced by a layer of limestone overlain by organic matter. The limestone could generate a large amount of alkalinity. Cell 1 will be able to oxidize and filter precipitates metals that are readily available before the AMD enters Cell 2. After the AMD becomes reduced and receives alkalinity, the effluent from cell two would then discharge into Cell 3. Cell 3 could oxidize and filter the remaining metal precipitates before discharging the water into the natural system.

Algae-Microbial Removal Of Manganese and Iron

At the Tennessee Valley Authority Fabius mine site an algal mat was used to remove iron and manganese from discharge from an oxidation pond before the drainage was put into a passive wetland (Phillips et al. 1994). The discharge from the oxidation pond was directed into three ponds: a pond with an algae mat consortium, a pond with limestone substrate, and a pond with pea gravel substrate. The microbial mat consisted of blue green algae and bacteria from the site cultured in the laboratory then returned to the site. A microbial mat slurry was broadcast over the experimental pond and fed with silage, grass clippings, added organic acids and several species of ensiling bacteria.

A comparison was made of the effectiveness of each pond to remove iron and manganese. The pond with algae mat consistently removed more iron and manganese than the other ponds even when the temperature dropped to 5 degrees centigrade in the winter (Phillips et al. 1994). A snowfall over the area in March of 1993 nearly destroyed the algal mat. Due to the susceptibility of the mat to winter conditions, treating acid mine drainage with algal mats in Montana does not seem to be a viable alternative.

Reverse Alkalinity Producing Systems

A pond can be constructed over a seep of AMD (Skousen et al. 1995). The pond is layered in the bottom with organic matter then overlain with limestone. The organic matter will reduce the AMD before it comes into contact with the limestone (Figure 5). The water can be ponded between 0.9 to 1.8 meters (3 to 6 feet) over the limestone-organic matter and can exit by a weir or a spillway. The effluent may then need to be aerated so hydrolysis and precipitation can occur. A site where this technology had been implemented could not be found in the literature.

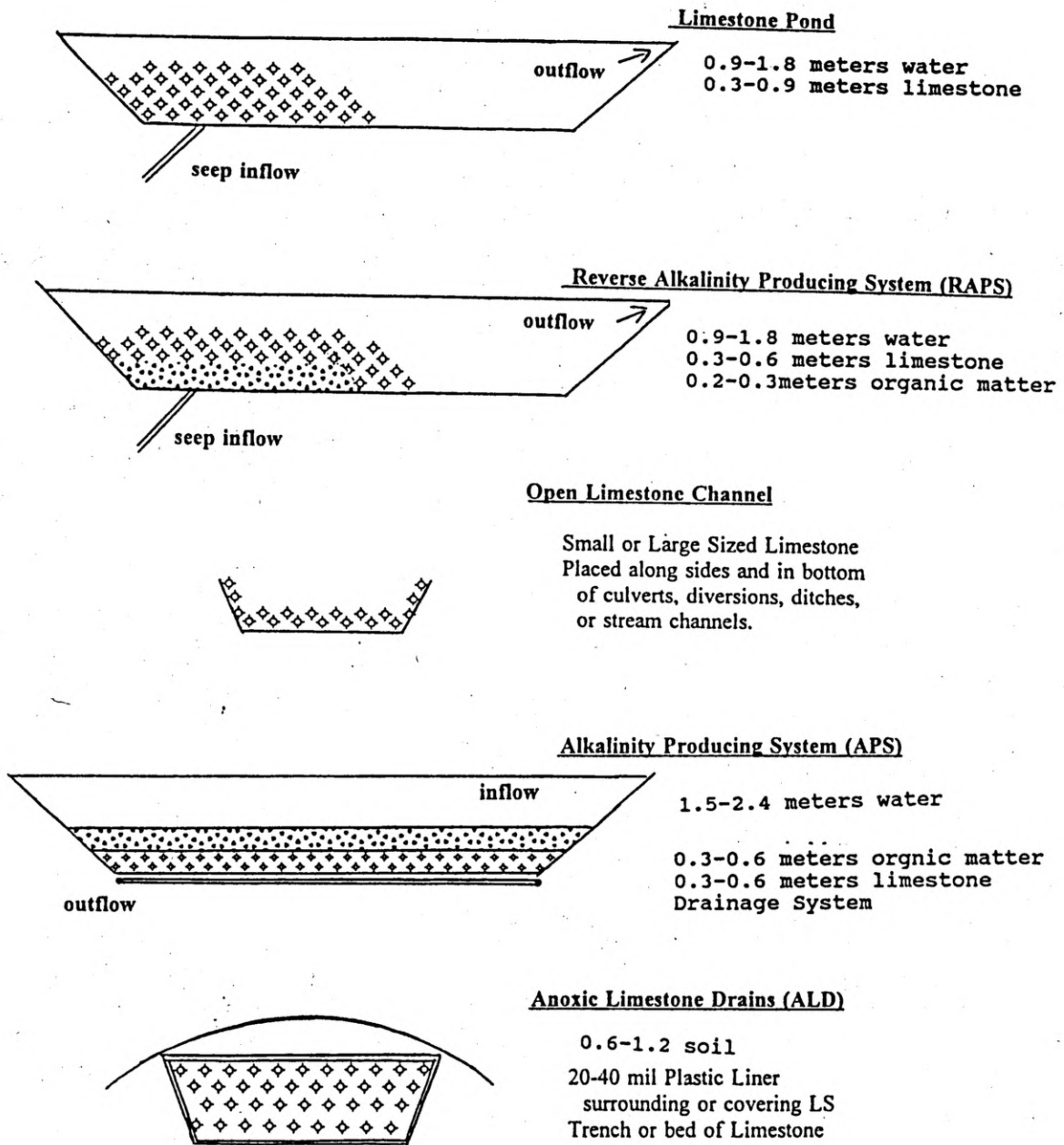


Figure 5. Schematic diagrams of passive treatment systems (Adapted Skousen et al. 1995).

Limestone Ponds

The idea of a limestone pond has been suggested for treating AMD seeps. Limestone is placed in the bottom of a pond constructed over a seep of AMD. The pond can be constructed to hold 1.2 to 3.0 (4 to 10 feet) of AMD with 0.3 to 0.9 meters (1 to 3 feet) of limestone in the bottom layer (Skousen et al. 1995). To ensure adequate limestone dissolution, the residence time should be 1 to 2 days. If coating of the limestone occurs by precipitates, a backhoe can be used to stir the limestone. As in the case of reverse sustaining alkalinity systems, a review of the literature a site could not be found where this idea had been implemented.

The idea of constructing a limestone pond may be experimented with at the Centerville "C" site. The catchment basin to the east could be converted into a limestone pond. However, research has not been conducted to determine the amount of alkalinity that can be generated by a pond or the amount of limestone required, or the size of the pond required to treat AMD. At present the pond is probably too small to provide enough alkalinity to pretreat the AMD. A cell at French Coulee and Centerville "C" could be converted into a limestone pond. However, without the reducing of the AMD before entering or coming in contact with the limestone, the limestone would soon become armored

and would require maintenance to sustain its performance. The larger size of the cells any provide a better opportunity to provide more alkalinity than the catchment pond, they may not provide enough without routine stirring and later replacement of the limestone.

CONCLUSION

The AMRB wetlands near Belt, MT are not performing adequately to produce effluent discharge that meets minimum standards. The wetlands do provide varying degrees of treatment. There are different reasons for the inadequate performance: winter conditions, inadequate size, extreme chemistry of the AMD, or a combination of all of these factors. It would be impossible to change Montana's winter weather and too expensive to redesign or reconstruct the wetland systems to improve their performance or increase their size. One possible solution is to treat the extreme chemistry of the AMD, so that influent chemistry can be more easily treated by the passive treatment systems. There are several processes that have been used in the eastern United States to treat or pretreat AMD. Most processes involve the addition of alkalinity to the drainage. By adding alkalinity to passive treatment systems, the effectiveness of the remediation of the AMD is improved. Alkalinity can be added through open limestone channels, sustaining alkalinity producing systems, anoxic limestone drains, limestone ponds, and reverse alkalinity producing systems. Some of these processes involve direct treatment of seeps and are not applicable to the study situations. The building of anoxic drains as pretreatment has failed at

Centerville and Stockett. The building of a post ALD to treat the anoxic discharge from the Centerville and French Coulee wetland would be risky due to past failure of ALDs at the sites. Both sites would require the construction of an additional settling pond. It would difficult to guarantee the French Coulee and Centerville substrate effluent would be sufficiently reduced or anoxic to allow an ALD to function without plugging. Also, tampering with the geotextile liners at French Coulee and Centerville would void the liners warranty. The remaining options of OLC and sustaining alkalinity producing systems remain viable options to treat AMD chemistry and bring the AMD chemical parameters into a range suitable for treatment with the wetlands. The open limestone channels option could be constructed at all three sites. A SAPS system is a viable option at the French Coulee and Centerville sites. A selection of several of these options could be constructed at each of the wetland systems to evaluate effectiveness to treat AMD.

LITERATURE CITED

- Brodie, G.A., 1990, Treatment of acid drainage using constructed wetlands experiences of the Tennessee Valley Authority. Proc. of the Nat. Symp. on Mining, Lexington, KY May 1990. University of Kentucky, Lexington, KY. pp 77-83.
- Brodie, G.A., Britt, C.R., Tomasewski, T.M., and Taylor, H.H., 1992, Anoxic drains to enhance performance of aerobic acid drainage treatment wetlands-Tennessee Valley Authority, West Virginia Surface Mine Task Force Symposium, Morgantown, West Virginia.
- Hedin, R., 1991, Constructing wetlands for treatment of acid mine drainage, U.S. Bureau of Mines, Pittsburgh Research Center, Pittsburgh, PA.
- Hedin, R., Watzlaf, G., Narin, R., 1994, Passive treatment of acid mine drainage with limestone, Journal of Environmental Quality, Vol 23, Nov-Dec, 1994, #6, pg 1338-1345.
- Kepler, D.A., McCleary, E.C., 1994, Successive alkalinity producing systems (SAPS) for the treatment of acidic mine drainage, in International Land Reclamation and Mine Drainage Conference and Abatement of Acidic Drainage, Pittsburgh, P.A.
- Kepler, D.A., McCleary, E.C., 1995, Successive alkalinity-producing systems (SAPS), in Proceedings Sixteenth Annual West Virginia Surface Mine Drainage Task Force Symposium, Morgantown, West Virginia.
- Peccia and Assoc., 1992, Operations and mine plan Cascade mine drainage, ARMB Report # 91-001, Helena, Mt.
- Peccia and Assoc., 1993, Cascade mine drainage: final report, AMRB Project, # 91-001, Robert Peccia and Assoc, Helena, MT.
- Pearson, F.H., McDonnell, A.J., 1974, Chemical Kinetics of Neutralization of Acidic Water by Crushed Limestone, Proc. 18, Water Resources Problems Related to Mining, American Water Resources, pg. 85-98.

- Phillips, P., Bender, J., Simms, R., Rodriguez-Eaton, S., Britt, C., 1994, Manganses and iron removal form coal mine drainage by use of a green algae-microbial mat consortium, International Land Reclamation and Mine Drainage Conference on the Abatement of Acidic Drainage, Pittsburgh, PA.
- Schafer and Assoc., 1994, Wetland status summary report, AMRB Report 93-007-05, Schafer and Assoc., Bozeman, MT.
- Skousen, J., 1991, Anoxic limestone drains for acid mine drainage treatment, Greenlands, Vol 21 (4), pp. 30-35.
- Skousen, J., Faulkner, B., 1992, Preliminary results of anoxic limestone drains in West Virginia, West Virginia Surface Mine Drainage Task Force Symposium, April 8-9, 1992, Morgantown, West Virginia.
- Skousen, J., Sextone, A., Garbutt, K., and Sencindiver, J., 1992, Wetlands for treating acid mine drainage, Greenlands, Vol 22, #4, pg.31-39.
- Skousen, J., Faulkner, B., and Sterner, P., 1995, Passive treatment and improvement of water quality, Proceedings Sixteenth Annual West Virginia Surface Mine Drainage Task Force Symposium.
- Stumm, W., and Morgan, J., 1970, Aquatic chemistry. Wiley New York. 583 p.
- Wildeman, T., Brodie, G., Gusek, J., 1993, Wetland design for mining operations Handbook for constructed wetlands receiving acid mine drainage, Colorado School of Mines, Golden, CO.
- Ziemkiewicz, P., Skousen, J., Lovett, R., 1994, Open limestone channels for testing acid mine drainage: A new look at an old idea, Greenlands, Vol 24, #4, pg. 36-41.