Environmental Impacts of Acid Mine Drainage in the Appalachian Region

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Abstract

Coal mining in the Appalachian region was not regulated with respect to environmental effects until the 1960's and 1970's. The lack of appropriate mining impact regulations and management protocols caused detrimental effects to the environment, natural resources, wildlife, and residents within the region. As time progressed, Appalachia residents, scientists, mining companies and many others started to notice dramatic changes in the environment within the region due to active mining operations and abandoned coal mine-lands. Acid mine drainage (AMD) was discovered and has since been well-documented for its acidic and metalliferous properties. AMD is a harmful water pollutant caused by the oxidation of pyrite and other sulfides which become exposed to surface conditions during mining operations. The pyrite, or other sulfur-containing minerals, react with surface and shallow subsurface water and oxygen causing a chemical reaction which results in sulfuric acid. The AMD then enters rivers and streams where it affects aquatic species, wildlife, humans, and the pH of waters. By the 1960's, altered ecosystems were observed due to mining overburden (spoil) disposal methods such as the "shoot and shove" method of contour surface (strip) mining. These methods impaired natural resources by contaminating water sources with acidity, heavy metals and bulk salts. As mining operations progressed in the pre-regulated era, mined lands were often abandoned leaving sulfidic materials exposed leading to the release of associated metals such as Al, Cd, Cu, Ni, and Zn which led to extreme water quality issues as toxicity became notable in contaminated streams causing teratogenic effects on fish and wildlife populations. Later, the Surface Mining Control and Reclamation Act of 1977 (SMCRA) was created to guide and regulate the environmental impacts of active and past coal mining operations. In particular, water quality protections implemented by SMCRA have made a significant improvement to the overall health and restoration of both active and abandoned mine lands (AML) in the Appalachian region.

Introduction

The Appalachian coalfields are one of the most diverse and unique regions in the United States and have supplied approximately 50 billion tons of marketed coal over the last two centuries (Zipper et al., 2021). This region extends from northwestern Pennsylvania south to northern Alabama, covering seven states of the eastern USA. (Zipper & Skousen, 2021a,b). Because of coal mining, the Appalachian region has faced many challenging environmental impacts due to the highly productive coal industry generating significant changes to the ecosystems, landscapes, and associated water quality. In particular, frequent exposure of reactive sulfidic minerals and associated heavy metals during mining operations caused many of these changes, along with poorly controlled methods of overlying rock overburden disposal. Much of the overburden was pushed into adjacent hillslopes and valleys using heavy mining equipment that was large enough to move considerable amounts of earthen materials. As a result, AMD became a byproduct in the form of discharge seeping from deep mines and leachates from coal refuse disposal areas which continues to be one of the most persistent environmental impacts caused by the mining industry prior to the SMCRA. However, the focal point of this research paper is to address the impacts from surface mining on the environment and current reclamation practices used to treat, improve and mitigate those impacts.

Coal mining became important for human advancements over time and so did environmental protection laws and regulations for the mining industry. Coal became an essential source of power in the industrial revolution in the 18th and 19th centuries and boosted the economy by providing jobs and lowering the cost of energy. In 1939, West Virginia (WV) was the first state to legislate the control of surface mining. Later, the WV Department of Mines was established as a regulatory agency enforcing reclamation of mined lands. By 1945, the Department of Mines required mine operators to pay $200 per acre for surety bonds which were intended to ensure that reclamation practices were completed. Additional laws required a $50 registration fee to be paid for each mining operation. Other Appalachian states followed and received authority to revoke mining permits when necessary. Years later, AML funds were created for prescribed reclamation, inspections, construction, maintenance of mined lands, and damages that affected neighboring landowners through coal mining. Following the establishment of widely variable state regulations, the federal Surface Mining Control and Reclamation Act was enacted in 1977 to unify national reclamation and environmental standards for coal mined lands (Skousen & Zipper, 2013). SMCRA and the Clean Water Act (CWA) of 1972 imposed requirements for the control and treatment of AMD by requiring each mining operation to acquire a National Pollutant Discharge Elimination System (NPDES) permit under the CWA (West Virginia University Extension, 2020).

Soon after the active coal mining era began in the 1940's, AMD became a major concern for many lifeforms in the Appalachian region. AMD was visually well documented by the1960's at which time an estimated 16,920 km of streams in Appalachia were impacted by AMD and other non-acidic mine drainages. However, not all Appalachian states had extensive inventories which made it difficult to determine a total of stream lengths impacted by AMD (Daniels et al., 2021). Many streams in Appalachia were surveyed to identify the concentrations of acidity, metals and their impacts to biotic diversity. In addition, the federally mandated (see OSMRE, 2019) Abandoned Mine Land Inventory System (AMLIS) documented AML prior to the enactment of SMCRA. During inventories, the AMLIS determined the risk of human health, safety, and welfare, as well as aquatic life and ecological health associated with AML and discharges. An estimated 7600 hydrologic units in Appalachia contain water that does not meet the standards for human consumption or agricultural and industrial use (Daniels et al., 2021). Below, Table 1 provides an overview of AMD-impacted stream miles within the Appalachian region and indicates the overall degree of impact.

Table 1: Inventoried units impacted by coal mine drainage from AML in Appalachian states and their current reclamation status. Units are stream miles. Funded and Unfunded columns indicate the number of stream miles "funded" or "unfunded" by the Abandoned Mine Reclamation Fund. Unfunded areas are yet to be remediated. Data from OSMRE (2019); adapted from Daniels et al., (2021).

State	Completed (Miles)	Funded (Miles)	Unfunded (Miles)	Total (Miles)
Kentucky	19,146	978	990	21,114
Maryland	268	1.0	2067	2336
Ohio	396.1	1.0	22	419.1
Pennsylvania	3589.7	276.9	150	4016.6

Tennessee	217	0.0	95	312
Virginia	7554	74	536	8164
West Virginia	26,372.3	1175	1320	28,867.3
Total	57,543.1	2,505.9	5,180.0	65,229.0

Formation of Acid Mine Drainage

The formation of AMD is caused by oxidative weathering of pyrite and other sulfidic minerals when exposed to oxygen and water in areas containing little or no geologic alkaline buffering strata. Pyrite can occur as crystalline or framboidal structures, however, oxidation reactions occur faster with framboidal pyrite structures due to its larger reactive surface area (Daniels et al., 2021). AMD drainage can be further catalyzed by the presence of *Thiobacillus ferroxidans* bacteria that greatly accelerate reaction rates when the pH drops below 5.5. Resulting reaction precipitates include ferrous iron (Fe^{2+}), sulfate, and acidity or protons (H^{+}) (Gwenzi, 2021). Equation 1 below shows the first step in the oxidation of reduced sulfur (S^{2-} or S_2^{2-}); which results in ferrous iron (Fe^{2+}), sulfate, and hydrogen protons. In the second step of the reaction, Fe^{2+} reacts directly with oxygen and hydrogen which produce ferric iron (Fe^{3+}) as shown in Equation 2 and is pH dependent (Daniels et. al., 2021). The oxidation process in Equation 2 proceeds quite slowly and is considered the rate limiting step of the reaction, however, when *Thiobaccillus thiooxidans* (a sulfur oxidizing bacteria) and *Thiobaccillus feroxidans* (a bacterium that oxidizes Fe^{2+} and Fe^{3+}) are present, the reaction is catalyzed and greatly increases the rate of acid production (Geidel & Caruccio, 2000). Subsequently, Equation 3 involves the hydrolysis reactions of ferric iron and water which produces ferric hydroxide and additional acidity (Gwenzi, 2021). Consequently, Equation 3 results in turbidity from sediments (a.k.a. “yellow boy”) in streams causing the water to appear yellowish-orange to reddish-brown in color. However, when AMD forms from geological minerals that are low in Fe, the water may appear relatively clear having no visual indication of acidity (Daniels et al., 2021). Lastly, the $FeS_{2(s)}$ in Equation 4 can be directly oxidized by excess $Fe^{3+}_{(aq)}$, which further oxidizes and hydrolyzes, releasing more H^{+} (Geidel & Carcuccio, 2000).

$$FeS_{2(s)} + 3.5O_{2(g)} + H_2O \rightarrow Fe^{2+}_{(aq)} + 2SO_4^{2+}{}_{(aq)} + 2H^+_{(aq)} \quad (1)$$

$$Fe^{2+}_{(aq)} + 0.25O_{2(g)} + H^+_{(aq)} \rightarrow Fe^{3+}_{(aq)} + 0.5H_2O \quad (2)$$

$$Fe^{3+}_{(aq)} + 3H_2O \rightarrow Fe(OH)_{3\,(s)} + 3H^+_{(aq)} \quad (3)$$

$$FeS_{2(s)} + 14Fe^{3+}_{(aq)} + 8H_2O \rightarrow 15Fe^{2+}_{(aq)} + 2SO_4^{2+}{}_{(aq)} + 16H^+_{(aq)} \quad (4)$$

Overall, Equation 1- 4 can be summarized as the overall suite of geochemical processes which produces AMD via the generation of H^+ which accounts for the total acidity production and lowers spoil, soil and water pH. Additionally, the resulting acidic conditions are responsible for releasing other contaminants such as metals, metalloids, and rare earth elements which are also released into streams simultaneously during oxidation or through dissolution influenced by acidic conditions (Gwenzi, 2021).

Treatment of Acid Mine Drainage

AML reclamation and remediation requires complex treatment and management due to the exposure of pyrite and other reduced elements to oxygen and water resulting in AMD pollution. Remediation of AMD in the Appalachian coalfields is most commonly accomplished through active or passive treatment systems used to control focused point sources and reduce their adverse impacts. Rivers and streams affected by AMD can also be treated using more dispersed watershed-scale treatment systems (Daniels et al., 2021).

Active Treatment Systems

Active treatment systems for AMD require long-term, chemical treatment that neutralizes AMD, precipitates metals and increases water pH. Calcium or sodium-based alkaline reagents such as lime, NaOH or anhydrous NH^{3+} (particularly in WV) are used in active treatment systems to raise the pH and neutralize and precipitate acid soluble metals (Daniels et al., 2021). The addition of neutralizing agents to reduce acidity allows metals to precipitate from the water assuming sufficient reaction and settling time.

Active treatment systems generally include an inflow ditch, a chemical storage tank, an application control system, holding ponds for settling and collection of metal precipitates, and a discharge point. The discharge point is monitored for compliance with the NPDES water quality criteria (West Virginia University Extension, 2020). Most AMD treatment systems are commonly designed to apply alkaline reagents in liquid form (e.g. NaOH) to AMD waters which have flow-control valves to control the rate of reagent additions entering the water (Daniels et al., 2021). The rate of neutralizing chemicals is determined by multiplying the flow (gpm), the acidity of the AMD (mg/L), and a conversion factor of 0.0022 giving an estimate of total acidity to be neutralized. The product of this calculation results in the number of tons of acid requiring neutralization per year (tons/year); this product can also be expressed in calcium carbonate equivalent (CCE) form. Finally, the product expressed in tons/year can be further multiplied by a conversion factor for each chemical to determine the amount of neutralizing reagent needed (West Virginia University Extension, 2020)

During the monitoring process, pollutant discharge levels are regulated based on standards and associated water quality standards set by the U.S. Environmental Protection Agency (EPA). For active mine permits since SMCRA, the NPDES permit is nested within the mining permit which is jointly administered by the states and OSMRE. The U.S. EPA pollutant discharge standards can be technology based or water quality based. Water quality discharge standards are typically used when discharge is released into streams with designated uses. If AMD discharge enters an impaired stream with certain designated uses, a Total Maximum Daily Load (TMDL) study is developed for that stream and load allocations for the mix of discharges within that watershed are determined to mitigate water quality issues (West Virginia University Extension, 2020).

Passive Treatment Systems

Passive treatment systems include a variety of natural biological, geochemical, and physical processes used to neutralize acidity and to first oxidize any soluble Fe^{2+} to the ferric form to allow for the acidity to be neutralized and then often to reduce it again to Fe^{2+}, and precipitate metals from mine waters. Passive treatment technologies include constructed wetlands, anoxic limestone drains (ALD), successive alkalinity producing systems (SAPS), limestone ponds, and open limestone channels (OLC) (Skousen et al., 2017). Contaminated waters can be treated via passive methods using artificial wetlands. Artificial wetlands function to sequester acid forming materials in a given area through microbial action which creates an anaerobic environment preventing additional formation of sulfuric acid and removing acidity via various pathways (AGI, 2022). Passive treatment systems are designed to function similarly to natural wetlands or other native biogeochemical processes (Daniels et al., 2021) and are designed based on elemental concentrations and flow volume, and land availability (Zipper et al., 2011). These systems can be labor intensive and require some maintenance depending on the type of technology implemented, and are often very costly to install initially due to construction and infrastructure needs of the system. Routine maintenance tasks include opening and closing of valves, clearing beaver dams, and replacement of alkaline or organic substrates (Daniels et al., 2021). However, the economic returns of passive systems often outweigh the initial costs since they can be conveniently installed in areas where active treatment systems would not be practical. They also alleviate the need for frequent personnel visits to adjust reagent additions and check on actively operating mechanical equipment.

Passive treatment systems are categorized into two types of systems: biological systems and geochemical systems. Passive treatment system designs are based on the flow or discharge of the water, the concentration of acid forming materials in the water, and the overall water quality discharge criteria. Depending on design conditions and intended function, one or more treatment systems may be installed as a combination of differing inline treatment processes (e.g. an aerobic surface wetland followed by a SAPS). Overall, land availability and site characteristics greatly influence the design and selection of passive treatment systems for AMD remediation (Zipper et al., 2011).

Biological treatment systems are constructed to mimic the functions of natural ecosystems. Biological systems include aerobic and anaerobic wetlands (AeWs and AnWs), Mn removal beds (MRBs), sulfate-reducing bioreactors (SRB), and vertical flow wetlands (VFWs). During the late 1970's and 1980's researchers discovered a correlation between the naturally-occurring *Sphagnum* bogs and *Typha* wetlands and the amelioration of AMD in Ohio and West Virginia as mine waters passed through the naturally existing bogs and swamps. However, it was also noted that over time, AMD eventually degraded the structure and function of many wetlands and their existing wetland plants (Skousen et al., 2017).

Aerobic wetlands are designed as shallow depressions or just a shallow basin filled with aquatic vegetation and loose substrate (Skousen et al., 2017). These are usually created to treat mildly acidic AMD or net-alkaline waters with slightly elevated Fe concentrations, and to provide residence time for Fe-oxidation allowing metals to precipitate. Anaerobic wetlands function to aerate the AMD waters flowing over the vegetation within the system which allows Fe-oxidation, hydrolysis, and settling of metal hydroxide flocs (Zipper et al., 2011; Skousen et al. 2017). However, effluent waters from an AeW are sometimes more acidic than influent waters requiring additions of limestone to the system to enhance its long-term efficacy (Skousen et al., 2017). **Figure 1** adapted from Zipper et al. (2011) exhibits a cross section view of a constructed aerobic wetland used for treating mildly acidic or net-alkaline waters. Within this system, aquatic vegetation such as *Typha* (cattails) is planted to allow translocation of O_2 to subsurface roots which further promotes oxidation of metals. Aquatic vegetation also reduces the velocity of water passing through the system which prevents erosion and channelization, and improves sedimentation of solid-phase metals.

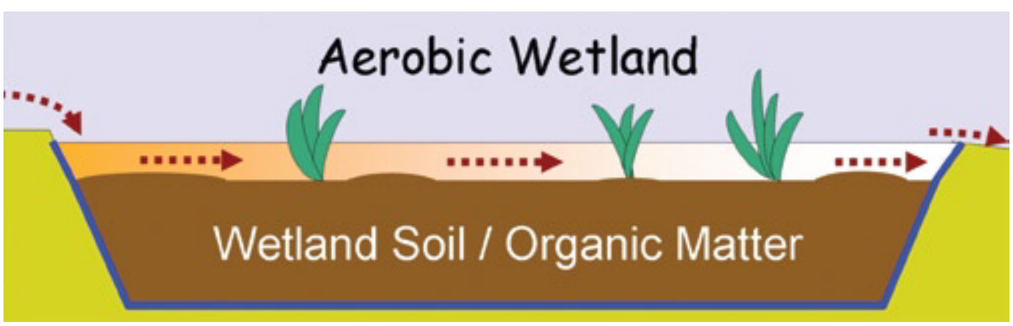

Figure 1. Cross-sectional view of an aerobic wetland. Aerobic wetlands are used to treat mildly acidic, circumneutral, or alkaline water containing dissolved metals such as Fe which are transformed into a solid phase by oxidation in AeWs. Adapted from Zipper et al. (2011).

On the other hand, simple surface flow driven AnWs are slightly different from AeWs having *Typha* and other aquatic vegetation planted in permeable substrates with ponded depths >30 cm which stimulates plant growth, reduces flow channelization, and adds fresh organic material to the substrate. The soil within an AnW system is mixed with peat moss, spent mushroom compost, sawdust, hay bales, manure, and other organic materials (Skousen et al., 2017). Limestone is either incorporated into the organic materials or added as bed beneath the organic matter as illustrated in **Figure 2** below; adopted from Skousen et al. (2011).

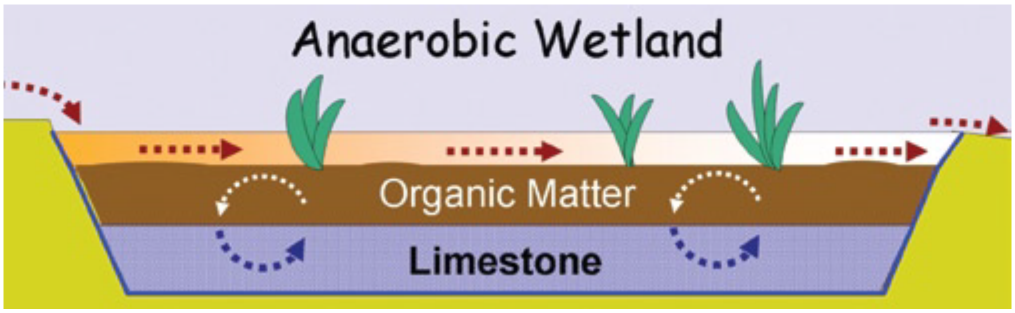

Figure 2. Cross-sectional view of an anaerobic wetland. Anaerobic wetlands are used to generate alkalinity via diffusion of treatment waters into and through substrate layers composed of organic materials placed over a layer of limestone. Adapted from Zipper et al. (2011).

Anaerobic wetlands are typically used to treat a lower volume of AMD flow and require a larger surface area to efficiently remove metals such as Fe, Al and Mn (Zipper et al., 2011). The limestone within AnWs generates alkalinity through carbonate dissolution along with some also generated by microbial sulfate reduction (Skousen et al., 2017) which occurs under anoxic conditions in the presence of sulfates and biodegradable organic material (Zipper et al., 2011). During the sulfate reduction process, influent sulfate-S is transformed into either hydrogen sulfide gas (H_2S) or a solid-phase precipitated reduced sulfides. In addition, sulfate reduction is very visible due to the presence of emerging bubbles from the substrate which creates a rotten egg odor as H_2S gas is released (Zipper et. al., 2011). It is important to note that in order for these types of anaerobic wetlands to effectively function, they need to be designed to ensure that influent water does have sufficient residence time to infiltrate into and interact with the compost and limestone layers at depth.

Mn removal beds (MRBs) are another form of passive treatment used on AMLs. MRBs are formed using enough limestone rock to create a large surface area as a growth substrate for specific bacteria via their respiration of organic matter which is used as their source of energy.

The functionality of MRBs relies on the presence of dissolved oxygen to promote oxidation and the addition of alkalinity producing materials to increase the pH to near circumneutral levels. Overall, MRBs are commonly used as a final step in passive mine water treatment (Skousen et al., 2017) since Mn often remains in effluent for longer periods of time after Fe and sulfate have been removed.

Sulfate-reducing bioreactors (SRB) are vertical flow compost-based treatment systems constructed using organic materials such as wood chips, straw, manures, oyster shells, etc. mixed with limestone creating a sulfate-reducing environment (Daniels et al., 2021). Sulfate-reducing bioreactors are designed for slow flow rates and used to treat very acidic AMD and metal-rich waters. These systems are most commonly used when selenium (Se) removal is also necessary via biological reduction (Skousen et. al., 2017). Selenium (Se) is typically found in fresh rock spoil and coal materials in the form of selenide and rapidly released during weathering. Selenite is unstable and converts to selenate during oxidation which occurs in oxic surface waters where it becomes very mobile in the environment (Skousen & Zipper, 2013). Similar to simple AnWs, these function through microbial activity consuming influent drainage O_2 within the organic material to create anoxic conditions enabling sulfate-reducing bacteria to produce alkalinity and remove metals from the water (Daniels et al., 2021). However, when O_2 are depleted, facultative and obligate anaerobes start using Fe, Mn, and other elements as terminal electron acceptors. The monosulfide FeS is also removed from the solution, is relatively short-lived, and precipitates into the organic material. However, if FeS is present in the effluent and settling pond, it needs to be retained in an anoxic environment to prevent re-oxidation and the formation of more acidity (Skousen et. al., 2017) since it is much more reactive than pyrite.

When SRBs are first created they are often filled with fresh water while the AMD waters are gradually introduced to initiate sulfate and Fe-reduction allowing the microbial system to equilibrate and begin functioning. Occasionally, SRBs are inoculated with microbes to advance a newly formed treatment system and increase the overall efficiency of the system (Skousen et al., 2017). These systems can be designed for use in a series of tanks or in large pond systems where the flow rate can be adjusted to ensure sulfide removal is successful (Daniels et al., 2021).

Vertical flow wetlands (VFWs), also known as successive alkalinity producing systems (SAPS) or reducing and alkalinity producing systems, and were initially developed in the 1980's (Skousen et al., 2017). These systems are designed with three major components consisting of an organic layer over a limestone layer with an underlying drainage system (Zipper et al., 2011) that pulls water down through the beds from the ponded surface. The limestone layer is between 0.5-1 m thick with a porous layer of filter fabric over it. An organic layer is then placed over the filter fabric which is usually 0.2-0.6 m in thickness. Finally, acidic water is ponded over these two layers at approximately 1-2 m in depth. As acidic water passes down through the organic layer, O_2 is consumed by the organic substrate and Fe is reduced to Fe^{2+}, and acid is neutralized to some extent in the organic layer through sulfate reduction. Then, the water passes into the perforated drainage system below the limestone layer and into an aerobic wetland or settling pond where any remaining Fe and Mn reoxidize and precipitate or are lime treated to force precipitation. Studies show that VFW treatment systems can be very effective for treating AMD, having the ability to reduce acidity rates ranging from 2 to 800 g m^{-2} day^{-1} with an average of 87 g m^{-2} day^{-1} (Skousen et al., 2017).

Other geochemical treatment systems are briefly described below and include anoxic limestone drains (ALDs), open limestone channels (OLCs), limestone leach beds (LLBs), steel slag leach beds (SSLBs), diversion wells, limestone sand (LS) treatment and Low-pH Fe oxidation channels. Geochemical treatment systems incorporate inorganic materials such as carbonates to reduce acidity in AMD waters.

Anoxic limestone drains (ALDs) were first constructed in the late 1980's to be used as a pre-treatment system for waters flowing into constructed wetlands. By the early 1990's, ALDs were observed improving the function of wetlands by meeting the effluent limitations without chemical treatment and have since been constructed as stand-alone systems where AMD discharges from deep mine portals. In order for the ALD to function properly over time the ALD must be sealed to prevent O_2 entering and keep CO_2 from escaping which helps the system to remain in an anaerobic condition. However, if the ALD does not maintain anaerobic conditions, the limestone in ALDs tends to clog when significant amounts of Fe^{3+} and Al^{3+} are present in the discharge which shortens the lifespan and functionality of the ALD (Skousen, 1997).

Open limestone channels (OLCs) are open channels or ditches filled with large limestone, approximately 8 cm or greater in size (Ziemkiewicz et al., 2003; Daniels et al., 2021). These systems can be used alone or in combination with other passive treatment systems such as settling ponds or wetlands placed at intermediate points for removal of precipitates (Ziemkiewicz et al., 1997; 2003). These systems are designed to be most effective on slopes greater than 12% with a high velocity and residence time (Ziemkiewicz et al., 2003). At high velocities, metal hydroxides are held in suspension preventing the clogging of limestone packing voids (Ziemkiewicz et al., 1997) and limiting formation of Fe-oxide coatings (armoring) on the rocks. Residence time is essential for adequate alkalinity production in OLCs; however, it creates armored limestone and reduces the rate of alkalinity production. To reduce armoring of limestone in OLCs, the system can be designed with a sloping grade to allow suspended particles to dislodge via the turbulence of flowing waters (Daniels et al., 2021).

Limestone leach beds (LLBs) are relatively easy to construct and receive a lot of use due to their low cost. They are designed as small basins filled with coarse limestone and receive AMD waters as influent water from AMD seeps or underground mine discharge which provides about 30 minutes of residence time. Studies found that a residence time of at least 30 minutes removed approximately 50% of the acid load from slightly acidic waters (Skousen et al., 2017).

Steel slag leach beds (SSLBs) are constructed using metal-silicates from old slag piles and grinding it into sand and fine gravel sized particles. This is known as a cost-effective approach for generating alkalinity that can be introduced into AMD discharge (Skousen et al., 2017). The use of SSLBs depend on the chemistry of AMD and the reactivity of the slag used. If AMD is low in metal content, SSLBs can be used by allowing the AMD discharge to flow through the SSLB to neutralize the acidity (Daniels et al., 2021). In a study by Kruse et al. (2019), SSLBs were found to effectively restore the mouth of the East Branch of Raccoon Creek, Ohio, to circumneutral pH, and were thought to potentially increase the recovery scores of macroinvertebrate species by improving aqueous and sediment conditions.

Diversion wells are another mechanism used for AMD treatment. Diversion wells are cylindrical concrete or metal tanks filled with limestone with a metal pipe extending down the length of the tank which carries AMD to the bottom of the tank. The AMD is carried under pressure through the metal discharge pipe which agitates and fluidizes the limestone in the tank thereby preventing coating and enhancing dissolution. Dissolution of limestone within this system aids in removal of Fe-hydroxide coatings and supposedly provides a continual limestone surface (Skousen et al., 2017). Limestone sand (LS) treatment is a very common method of reducing acidity on a watershed scale and is largely used by the State of West Virginia to restore many river systems affected by AMD deposition. LS is applied directly to the stream and suspended by streamflow which transports and redistributes the sand downstream while neutralizing acidity. LS treatment requires quarterly additions of LS to maintain water quality (Skousen et al., 2017).

Lastly, low-pH Fe oxidation channels can be used as a passive treatment system for high Fe discharges. This treatment system is relatively new and is designed as a shallow channel lined with limestone. Similarly to other treatment systems, low-pH Fe oxidation channels cause Fe to oxidize and precipitate onto the limestone rocks. However, these systems are different from OLCs since they can also be constructed with sandstone aggregate and the system promotes adsorption and co-precipitation of Fe-hydroxides onto rocks jointly with the presences of Fe-oxidizing bacteria. This system is pH dependent, so at a pH above 4.5, Fe^{2+} oxidation increases with abiotic and biotic catalysis. However, if the pH is lower than 4.5, oxidation is catalyzed by specialized bacteria. Overall, low-pH Fe oxidation channels are still being tested and some research is still being conducted to determine the efficiency and long-term effectiveness of the treatment method (Skousen et al., 2017).

Impacts to Humans

As discussed above, water quality in Appalachia was greatly affected by AMD resulting in stream pollution, but according to some authors, it has also affected human health (Gohlke, 2021). Acid mine drainage leaching from AML into streams and rivers caused an decrease in pH, and consequently, toxicity of waters within the Appalachian region leading to unsafe drinking conditions for humans. Stream contaminants caused by mining activities degraded drinking water sources and increased rates of chronic disease among residents in coalfield communities in Appalachia. Some studies within the Appalachian region revealed an occurrence of gastrointestinal cancers (stomach, lip, mouth, throat) due to stream pollution from coal mining (Krometis et al., 2017). Recent research identified a correlation directly linking the degradation of invertebrate health directly to human health, therefore, a positive correlation between average stream health scores (WV Stream Condition Indices, or WVSCI) and the incidence of digestive, breast, respiratory and urinary cancer rates at the county level in West Virginia (Krometis et al., 2017). These human health conditions were also assumed to have been associated due to exposure of oxidized metals and nonmetallic ions (e.g. As and Se) released from weathering overburden (Krometis et al., 2017), materials which cause AMD.

Although some evidence is lacking to directly correlate AMD with health disparities in Appalachia, it is clear coal mining has been associated with health risks; and some which resulted in mortality. A study by Hendryx and Ahern (2008) found that West Virginians who lived within close proximity of heavy coal production sites were more likely to have a worsened health status and have a higher risk of cardiopulmonary disease, lung disease, cardiovascular disease, diabetes, and kidney disease. More specifically, diseases such as chronic obstructive pulmonary disorder (COPD), black lung disease, and hypertension were found to be most commonly associated with coal mining; cardiopulmonary disease was the most notable among these conditions. The overall findings from this study concluded that these conditions were mostly associated with the occupational hazards of coal mining activities such as air pollution from burning coal, health risks associated with carbon dioxide, and community exposure to coal mining sites (Hendryx & Ahern, 2008). Another study by Almberg et al. (2018) was focused on progressive massive fibrosis (PMF), also known as mass-like conglomerates in the upper pulmonary lobes, and revealed an increased number of coal miners who filed for black lung benefits from 1970 to 2016. This study identified an increase in the number of Black Lung Program benefits in 1996, resulting in 2,474 cases of PMF from 1996 to 2016. Findings from this study also reported the majority of these cases were from coal miners working in the central Appalachian states of Kentucky, Virginia, and West Virginia (Almberg et al., 2018). While these studies identify health risks associated with coal mining, no hard evidence was found revealing a direct correlation between AMD and health disparities in association with coal mining.

Impacts to Aquatic Ecosystems

The Appalachian region contains the most biologically diverse freshwater fauna in the world. Rivers and streams in Appalachia harbor the richest species of crayfish, mussels, salamanders, amphibians, fish, benthic macroinvertebrates, and aquatic plant species (Merovich et al., 2021). Many of these aquatic species have become threatened due to anthropogenic pollutants such as AMD and sedimentation from mining. AMD has caused a reduction in fish and macroinvertibrate taxa, such as mayflies (*Ephemeroptera*), due to stressors such as low pH, reduced nutrient availability, and direct toxicity within rivers and streams (Zipper & Skousen, 2021b).

AMD-impacted streams are often observed with slower carbon turnover and associated nutrient cycling due to decreased decomposition of organic matter by macroinvertebrate shredders, filterers, and scrapers (Daniels et al., 2021). A study associated with impacts of AMD on stream ecosystem structure and function was conducted by Bott et al. (2012) who discovered a reduction in total macroinvertebrate density of each AMD polluted stream. Densities were reduced by 76% compared to reference streams receiving AMD, albeit with near-neutral in-stream pH, and a slightly acid reference stream (Bott et al. 2012; Daniels et al. 2021). In addition, pollution-sensitive taxa populations such as *Emphemeroptera, Trichoptera*, and *Plecoptera* were reduced by 96% in streams containing AMD discharge compared to reference streams (Bott et al. 2012). Bott et al. concluded that direct effects of AMD are most often observed at a local scale where $Fe(OH)_2$ precipitation visibly degrades aquatic habitats and reduces macroinvertebrate taxa populations, and promotes algae growth. Overall, AMD can decrease water pH to 3.0 or below causing an elevation in Fe, Al, and Mn, and trace elements resulting in toxic conditions, thereby altering water quality and decreasing biodiversity in rivers and streams (Zipper & Skousen, 2021b).

Impacts to Terrestrial Ecosystems

Wildlife and other semi-aquatic terrestrial species have been directly and indirectly affected by AMD as receiving rivers and streams served as water supply sources. Wildlife habitat loss in Appalachian coalfields devastated many terrestrial species as forested lands were converted to grasslands. The lack of reforestation on post-mined lands caused a reduction of forest cover and food supply. Although AMD is not the direct cause of the decline in wildlife biodiversity, coal mining interrupted the life cycles and habitats within the Appalachian region. For example, the taxonomic richness of salamander populations in Appalachian coalfields were reduced where habitats were adjacent to streams emerging from valley fills (Zipper & Skousen, 2021b).

Summary and Conclusions

Coal mining in the Appalachian region was a booming industry during the early years (late 1800's to mid- 1990's) and the harmful effects that future generations would face were not considered until environmental concerns became notable. Fortunately, today's mining laws and regulations ensure permit compliance requiring all mining sites are returned to approximate pre-mining conditions and appropriate land uses. In addition, SMCRA regulations require water monitoring data to be documented during the permitting process to indicate baseline conditions for a given area; monitoring data must include flow rate, pH, Fe, and Mn concentrations, and total suspended solids (Skousen & Zipper, 2013). Water quality criteria must be met for groundwater and surface water discharges during active mining and for at least five years following final reclamation. In addition, the CWA also requires mining permits which limits pollutant discharge into rivers and streams to help maintain water quality standards (Zipper & Skousen, 2021b).

Through research and development of active and passive treatment systems, water quality in rivers and streams throughout the Appalachian region has improved; some to the point of meeting regulatory targets for desired macroinvertebrates and fish populations (Daniels et al., 2021).

Since the establishment of the SMCRA, the effects of AMD from AML have been reduced; now new mining operations are not causing AMD discharge (Daniels et al., 2021).

Efforts to restore AMLs within the region are still ongoing today, but would not be as successful without the SMCRA, the CWA, and other regulatory agencies who oversee mining reclamation and environmental quality concerns.

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