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Assessment of Risks and Benefits for Pennsylvania Water Sources When Utilizing Acid Mine Drainage for Hydraulic Fracturing

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Assessment of Risks and Benefits for Pennsylvania Water Sources When Utilizing Acid Mine Drainage for Hydraulic Fracturing

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1.0 - Introduction

This research assesses the potential environmental risks and benefits to Pennsylvania water sources when utilizing acid mine drainage for hydraulic fracturing operations. This chapter presents background information on meeting U.S. energy demand through new technologies such as the use of hydraulic fracturing in the State of Pennsylvania.

1.1 Energy Demand and Hydraulic Fracturing

Energy demand within the United States continues to grow. The reason for this growth is economic expansion, which traditionally remains the primary political concern among registered voters in the United States (Newport 2014). Harnessing energy is essential for transportation, agriculture, real estate development, technological research, and manufacturing which are key drivers in the United States economy (Bureau of Economic Analysis 2015). Since the end of the Cold War and the rise of globalization, competition for economic growth has led to an increase in global energy demand. The increase of global energy demand has a positive correlation with energy cost (Government Accountability Office 2007).

The cost of specific energy sources could also include irreversible damage absorbed by humans, ecosystems, and property (USGS 2015). Scientific research from recent decades has generated concern among environmental professionals and stakeholders over the impacts of utilizing specific energy sources. Degradation of the atmosphere, soil, and waterways pose potential threats to environmental health, the health of U.S. citizens, and the economy (WHO 2015)

The increase of energy demand creates opportunities for energy companies to generate more supply in the market for international consumers. In order to increase supply, new energy reserves must be identified and extracted. Energy reserves can be in the form of sunlight, water, cell division, or fossil fuels. Fossil fuels have traditionally been the most common source of energy for the modern day economy. Demand for new sources of fossil fuels has resulted in energy companies exploring geological formations in perilous and obscure locations to identify these reserves (EIA 2014).

Historically the most commonly used form of fossil fuel in the United States has been coal. Coal powered the industrial revolution of the 19th century creating a carbon based industrial society. The United States has some of the largest coal reserves in the world providing a cheap and secure source of energy. To this day, coal provides the United States with over a third of its energy primarily through the generation of electricity (EIA 2015).

Concerns regarding the environmental costs of coal have been present since mining it began. Initially the most visible impacts to environmental health were identified by stakeholders located near mining and coal plants. These concerns were related to air quality, soil degradation, and waterway contamination (USGS 2015). Recent research suggests as one of the main sources of energy, coal is the most damaging to the atmosphere in the form of volatile organic compounds (VOCs). VOCs are reactive compounds driving climate change when present in the atmosphere. The increase of VOCs has motivated consumers and policy makers to look towards less impactful sources of energy such as natural gas (EPA 2015).

Natural gas has been an increasingly consumed form of energy in the United States. Historically, natural gas has been a by-product of oil extraction which is either burned off or stored in injection wells until market conditions make its sale profitable. The global energy demand has increased the profitability of natural gas over the past several decades. Natural gas has become increasingly viable in the current energy market such that extraction of shale natural gas is profitable (EIA 2015).   New forms of advanced technology are necessary for extraction of recently discovered natural gas and oil in shale formations. Horizontal directional drilling within bedrock has made the extraction of natural gas and oil within shale formations possible. This type of drilling is known as hydraulic fracturing or "fracking". Hydraulic fracturing involves vertical and horizontal drilling with the use of explosives and millions of gallons of water mixed with proppant (Healy 2012). The industrial phenomenon of hydraulic fracturing has significantly impacted U.S. society over the past decade. Hydraulic fracturing is an example of technology advancing at speeds preventing average citizens from fully understanding its function and potential impacts. This disconnect between technology capabilities and public knowledge fuels conflict and misunderstanding between stakeholders surrounding the issue of hydraulic fracturing. It is important to asses all current and potential impacts related to this influential technology that polarizes the public.

1.2 Pennsylvania Energy Sources and Industrial Impacts

Pennsylvania is historically an important source for discoveries and innovations in the energy sector. The reason for this is due to geological properties and background in Pennsylvania that have generated vast energy reserves in the form of coal, natural gas, and oil. Coal and oil have been extracted from Pennsylvania sub-surfaces on a mass scale since the early 19th century (Bertheaud and Pollman 2009). Access to energy reserves, waterways, and agricultural land made Pennsylvania a keystone state in the industrial revolution. The diversity of resources brought waves of immigrants and urban development resulting in one of the most populous and prosperous states (Bertheaud and Pollman 2009).

Currently Pennsylvania is the second leading state in natural gas exploration and extraction (EIA 2013). The industrial practice of fracking in Pennsylvania began in 2005. There are now over 7,000 active hydraulic fracturing wells registered in the state of Pennsylvania overlying the Marcellus Shale formation (PADEP 2015). This rapid change in the Pennsylvania landscape has brought immigrants and investment. These new forms of capital create employment and infrastructure opportunities for a state that has economically and industrially declined in recent decades (Considine et al. 2010). Accompanied with the potential of massive benefits is the potential for serious cost. Hydraulic fracturing has been practiced in Pennsylvania for only 10 years and the impacts of the industrial technique are still heavily contested.

Pennsylvania has experienced inexpensive accessible energy ever since the 19th century when it became a fossil fuel provider for the industrial revolution. The most visible and long lasting industrial cost for environmental health has been in the form of acid mine drainage. Acid mine drainage (AMD) is the acidification of waterways from the runoff of mines that have mostly been abandoned in Pennsylvania (Bertheaud and Pollman 2009). Waterways containing AMD will often have an unnatural orange and yellow coloring known as yellow boy. AMD impacted waterways contain high concentrations of heavy metals. The acid and heavy metals erode biodiversity in watersheds through the degradation of water quality and soil in watersheds. Currently Pennsylvania has 3,000 miles of AMD impacted streams. AMD costs the state of Pennsylvania $67 million a year in lost sports fishing revenue and it will cost $5 to $15 billion to restore the AMD impacted streams (USGS 2015).

Pennsylvania has over 83,000 miles of streams within its borders making it second behind Alaska in stream mileage (Pennsylvania Fish and Boat 2015). These streams support the dominant economic driver in the state which is agriculture. Pennsylvania has 62,000 family farms comprised of 7.7 million acres which annually contribute nearly $75 billion to the state economy (Pennsylvania Department of Agriculture 2015). The 83,000 miles of streams also support fishing related activities that generate more than $1.34 billion to the Pennsylvania economy every year (Pennsylvania Aquaculture Office 2015).

The waterways of Pennsylvania now support the multibillion dollar expanding hydraulic fracturing industry along the Marcellus Shale formation. Fracking is a very water intensive process requiring millions of gallons of water injected into the bedrock of the shale formation (Mantell 2011). Concerns from stakeholders that depend on the vast waterways have grown regarding water consumption, wastewater generation from hydraulically fracking flowback fluids, (HFFF) and potential contamination of watersheds. Stakeholders and environmental managers have looked to alternative sources of water that limit the impact of consumption and watershed risks (PADEP 2011).

One proposed source of water from watershed groups, hydraulic fracturing professionals, and scientists is the use of AMD that currently contaminates 3,000 miles of streams in Pennsylvania. This practice could potentially save hundreds of millions of water from Pennsylvania waterways. Stakeholders see an opportunity to provide relief to AMD impacted watersheds through the removal of AMD from surface waters (PADEP 2011). A recent study from the Nicholas School of Environmental Studies at Duke University found beneficial results from the use of AMD when mixed with HFFF. When the two are proportionally mixed, concentrations of heavy metals and radium decrease in the HFFF (Kondash et al 2013).

The use of AMD in the fracking process is legal and currently used by some hydraulic fracturing sites. Many hydraulic fracturing operations are hesitant to use AMD out of uncertain liability laws. The Pennsylvania State Legislature is debating whether to extend the Environmental Good Samaritan Act to the hydraulic fracturing industry to facilitate the use of AMD in the industrial practice (PADEP 2011). The Environmental Good Samaritan Act historically has been utilized to protect parties that assist in environmental remediation without profit or gain. The Pennsylvania Department of Environmental Protection has drafted a white paper proposing the use of AMD in the hydraulic fracturing process. Water quality parameters, storage facility requirements, and liability have been addressed in the state document (PADEP 2011).

1.3 Research Summary

The use of AMD in the hydraulic fracturing process increases the complexity of a polarizing topic. Many stakeholders see a potential to provide relief to multiple environmental problems while others see it as another threat to the Pennsylvania water supply so many parties depend on. This research will evaluate data from Pennsylvania state agencies and additional academic research regarding AMD and fracking to assess potential benefits, risks, and complications for Pennsylvania watersheds when using AMD in the fracking process.

This chapter presents background information on energy demand and hydraulic fracturing, as well as energy sources and industrial impacts within the state of Pennsylvania. Chapter 2 presents the background information on hydraulic fracturing and water sources to support the fracturing operations. Chapter 3 discusses the properties of acid mine drainage and its potential use as water supply for hydraulic fracturing. Chapter 4 discusses the use of acid mine drainage in the hydraulic fracturing operations in Pennsylvania. Chapter 5 presents research benefits, risks, challenges, conclusions and recommendations.

2.0 - Hydraulic Fracturing in Pennsylvania and Use of Water Sources

Pennsylvania has a history of supplying energy and innovations in the energy sector. Pennsylvania's geological properties have generated vast reserves of coal, natural gas, and oil. Coal and oil were primary energy sources extracted from Pennsylvania since the early 19th century (Bertheaud and Pollman 2009). Energy reserves, ocean access, and farming allowed Pennsylvania to play a dominant role in the industrial revolution. Economic opportunity from resources brought immigrants and urbanization creating a state with a high population and economic prosperity. (Bertheaud and Pollman 2009).

Pennsylvania is the second highest producer of natural gas behind Texas (EIA 2013). Since the beginning of hydraulic fracturing in 2005, there are now over 7,000 active hydraulic fracturing wells in the state of Pennsylvania (PADEP 2015). This rapid expansion within Pennsylvania has brought capital and professional transplants. These new resources create economic and development opportunities for communities that have been negatively impacted economically in recent decades. Potential benefits and cost are heavily contested since hydraulic fracturing has been practiced in Pennsylvania for only 10 years, making the impacts of this industrial technology unknown (Cosidine et al. 2010).

There are 7,000 active hydraulic fracturing wells in the state of Pennsylvania (PADEP 2015). Large sums of capital and resources have been invested in communities overlying the Marcellus Shale region in order to erect and operate hydraulic fracturing sites. Hydraulic fracturing operations are met with optimism and resistance due to economic benefits and environmental risks. For many communities that have been negatively impacted by contemporary economic trends, hydraulic fracturing is an industry that can potentially generate high paying jobs for underemployed residents. Local government stakeholders also see an opportunity for tax revenue that can repair or expand current societal infrastructure (Cosidine et al. 2010). Environmental risks are not entirely overlooked by stakeholders, but can be considered acceptable when economic benefits have strong potential.

2.1 Hydraulic Fracturing and Pennsylvania Economic Impacts

Cosidine et al (2010) estimated the total economic input from Marcellus Shale hydraulic fracturing operations in Pennsylvania at $7.17 billion as of 2009. Direct economic input was estimated at $7.7 billion which includes taxes, legal fees, and real estate transactions. Indirect economic inputs are estimated at $1.56 billion which are the services needed to construct and maintain fracking operations across the state (Cosidine et al. 2010). These services include subcontracts awarded to firms and the purchasing of materials. Induced economic impacts are estimated at $1.84 billion which are the consumer transactions of residents receiving financial benefits from direct and indirect inputs. 2020 predictions for economic input are projected to decline upstream with direct inputs. Indirect and induced economic impacts for 2020 are projected to increase in Pennsylvania to $18 billion generating $1.8 billion in tax revenue and over 20,000 jobs (Cosidine et al. 2010).

2. 2 Hydraulic Fracturing Mechanics

Hydraulic fracturing is a technique used by engineers to enhance the method of extracting oil and natural gas from shale bedrock containing hydrocarbon reservoirs. The purpose of hydraulic fracturing is to improve the permeability of the bedrock through pressure induced fractures (Healy 2012). This property of permeability is controlled through the pore fluid pressure and the in situ stress field which is the strength of the bedrock. Bedrock strength is dependent on multiple factors such as temperature, elasticity, and pore water pressure (Healy 2012).

Before the hydraulic fracturing process begins, a vertical well and a horizontal well must be drilled. Vertical drilling begins from the surface until contact with bedrock is made which on average is 7,500 feet (Trouba and Abeldt 2014). Once the drill reaches the bedrock it makes a 90 degree turn into the bedrock containing oil and or natural gas. The vertical drilling will go for a mile or longer. While the drilling is taking place, the well is reinforced with layers of steel and concrete to protect freshwater aquifers from contamination (Trouba and Abeldt 2014).

The fracturing process begins after the drilling of the wells. Fracturing involves the injection of a water diluted fluid known as proppant into a wellbore. The average Marcellus fracking well uses 5.6 million gallons for the initial formation (Mantell 2011). Fracking fluid is usually 85% water and 13% proppant (Fisher et al. 2013). Proppant contains quartz rich sand, ceramic pellets, and small incandescent particles (Healy 2012). When the fluid is pumped under high pressure into the well, it travels thousands of feet into the bedrock. The water will then change directions into a horizontal well extending thousands of feet where fractures from controlled explosions are present in the shale. The proppant in the fluid allows for the fractures to expand and remain open while oil and natural gas escape into the well (EPA 2014). This causes the bedrock shale encompassing the well to shatter. Fissures from the pressure allows the oil or natural gas to flow into the well where it is collected at the surface (Trouba and Abeldt 2014). Fractures within the bedrock are due to the in situ stress field which determines the strength of the bedrock (Healy 2012).

Three types of fractures that occur in the bedrock are tensile, shear, and a hybrid of the two. Shear fractures occur vertically from natural stress on the bedrock which can be enhanced or reactivated through hydraulic fracturing. Tensile fractures form when perpendicular pressure exceeds the strength of the bedrock (Healy 2012). Hydraulic fracturing does not control or decide which type of fracture occurs in the bedrock during the injection of water. Types of fractures are completely dependent upon geological properties of the bedrock where the hydraulic fracturing occurs (Healy 2012).

When pressure from the fluid is released, the oil and natural gas will flow to the surface where it is captured. On average 10-15% of the fluid returns to the surface of the Marcellus hydraulic fracturing wells which is classified as flowback (HFFF) (Mantell 2011). Fluid that returns to the surface due to pressure release within the first two weeks of fracking is HFFF, while fluids that return within two weeks are considered production fluids (Haluszczak et al. 2012). The HFFF is stored onsite in pits or tanks until it is treated, disposed, or recycled. HFFF usually contains proppant, radionuclides, hydrocarbons, brines, and metals (EPA 2004). If local regulations permit it, the HFFF can be treated at wastewater treatment plants and deposited in surface water bodies. HFFF can be recycled for future fracturing processes or disposed into injection wells. Injection wells deposit the HFFF hundreds or thousands of feet into bedrock below soil and groundwater (EPA 2004). HFFF is exempt from the Clean Water Act originally passed in 1974. The Energy Policy Act of 2005 excludes HFFF from drinking water standards from naturally occurring contaminants (Finkel and. Hays 2013).

2.3 Hydraulic Fracturing Water Source Impacts

HFFF constituents generate concern among private residents and public services. A study conducted by Haluszczak et al. (2013) found that HFFF had high concentrations of total dissolved solids (TDS), chlorine (Cl), Bromine (Br), Strontium (Sr), sodium (Na), calcium (Ca), barium (Ba), and radium (Ra). The concentration levels of Cl and TDS were 5-10 times higher than those of seawater (Haluszczak et al. 2013). The concentrations of Ba and Ra exceeded concentrations within drinking water compliance. In a study regarding anion concentrations from HFFF in Pennsylvania Marcellus Shale wells, Fisher et al. (2013) found concentrations of sulfate, bromide, and chloride. Concentrations of bromide and chloride increased 10-fold after the hydraulic fracturing process had been completed and proppant flowed back while sulfate concentrations remained the same.

Constituents of HFFF are of great concern to stakeholders because many of the contaminants are not easily removed through conventional wastewater treatment. One option for hydraulic fracturing operations in Pennsylvania to dispose of wastewater is through injecting wastewater volumes into deep wells (EPA 2004). When hydraulic fracturing wastewater is disposed of through conventional treatment plants, it can lead to surface water contamination. This is due to the remaining presence of contaminants when treated water is injected into surface waters (Warner et al. 2013)

The dominant concern among the public related to fracking operations is the impact on water bodies. Vulnerable water bodies include both surface waters and groundwater aquifers. Water bodies can be impacted through extraction and potential contamination. Most groundwater studies focus on the presence of methane in aquifers. Surface water studies tend to analyze the presence of HFFF constituents in water bodies.

The U.S. House of Representatives Committee of Energy and Commerce conducted an investigation in 2011 which identified that 14 major hydraulic fracturing companies used over 2500 proppant products containing 750 different chemicals (Entrekin et al. 2011). 29 of the identified chemicals are known carcinogens or highly toxic. Compounds included xylene, lead, formaldehyde, benzene, toluene, and ethylbenzene. Combined with the heavy metals and radionuclides of HFFF, these constituents accompanied with soil disturbances pose potential risks to nearby surface waters (Entrekin et al. 2011)

Warner et al. (2013) analyzed data from water samples in the Blackclick Creek of Pennsylvania near effluent streams of the Josephine Brine Treatment Facility. Concentrations of elements in the effluent and the surface water varied compared to the concentrations of the original HFFF. Elements Br and Cl values were similar to the concentration values of the HFFF before treatment which demonstrates the low impact of wastewater treatment with these elements. Ca, Sr, and Na concentrations had more varied values throughout the study proving that the treatment had some form of impact on these elements (Warner et al. 2013). Sulfate concentrations appeared to be enriched from the wastewater treatment possibly due to the additive of Na2SO4 during treatment. Ba and Ra concentrations of effluent samples showed a 99% decrease from HFFF concentrations proving the treatment process to be effective with removing the elements. Sr ratios and concentrations remained the same as the HFFF value before treatment showing little impact from the wastewater treatment process (Warner et al. 2013).

The surface water values downstream of the effluent discharge were similar to the concentrations of the effluent reading (Warner et al. 2013). Br and Cl had the highest enrichment factors compared to concentrations upstream from the wastewater facility. Cl had concentrations 2-10 times higher than the mean concentrations of compared western Pennsylvania streams. Na, Mg, Ba, Ca, and Sr did not show enrichment downstream of the wastewater facility which could possibly be due to the uptake of the elements through sediments (Warner et al 2013). Ra also appeared to have low concentrations in surface water downstream from the facility, but this is most likely due to the adsorption of Ra from sediments. This could be due to the high salinity values of the water quality near the treatment facility which enhances Ra adsorption. Ra poses a threat to benthic organisms and vegetation though the route of bioaccumulation (Warner et al. 2013).

Ferrar et al. (2013) conducted research comparing the HFFF and effluent concentrations on the Josephine Brine Treatment Facility along with two other wastewater treatment plants in Greene County, PA and Mckeesport, PA. The study measured the concentrations of Ba, magnesium (Mg), Cl, manganese (Mn), Ca, Br, Sr, benzene, sulfate, TDS, xylenes, ethylbenzene, toluene, 2-butoxyethanol, and turbidity. Most of the constituents had significant decreases after treatment from the wastewater facilities. The Greene County facility experienced a decrease in all constituents after HFFF had been treated. The Mckeesport facility experienced a decrease in all constituents except for little difference in Br concentrations and an increase in Mg and Ca. For the Josephine Brine Treatment Facility, Ferrar et al. (2013) found decreases in all constituents except for Mn, sulfate and 2-n-butoxyethanol. 2-n-butoxyethanol is known to have carcinogenic impacts which is a concern. All three had concentrations above the MCL for drinking water for Ba and Sr (recommended MCL for Sr). Benzene concentrations near the

Josephine Brine Treatment Facility were above the EPA and MCL human health criteria (Ferrar et al. 2013)

Soil disturbances and alterations provide potential contaminants an effective route into water bodies. Many of these changes in landscape from hydraulic fracturing operations are located in remote and relatively undisturbed regions of rural Pennsylvania. Runoff, erosion, and sedimentation degrade the quality of water bodies which potentially harms the environmental health of stakeholders (PADEP 2013). Soil disturbances in the form of erosion, sedimentation, and runoff are common routes for contaminants to enter water bodies. Hydraulic fracturing can enhance erosion and sedimentation within watersheds due to drilling and vehicles operating on sites. Operating hydraulic fracturing wells must use Best Management Practices when disturbing soil areas of 5,000 square feet or more in Pennsylvania. A major concern related to erosion, sedimentation, and runoff is eutrophication due to the presence of phosphates and nitrates (PADEP 2013).

Hydraulic fracturing case studies regarding soil disturbances have found correlations between hydraulic fracturing operations and water quality degradation of adjacent surface waters. Entrekin et al. (2011) found a positive correlation between well density in watersheds and turbidity among seven streams in the Fayetteville shale region of Oklahoma and Arkansas. Burton et al. (2014) conducted a study of 16 watersheds in the Fayetteville shale region monitoring soil and surface waters surrounding hydraulic fracturing operations. Burton et al. (2014) found a positive correlation with paved roads around hydraulic fracturing sites and the increase of conductivity in nearby surface waters. McBroom et al. (2012) conducted a study of hydraulic fracturing well runoff and sedimentation of the Alto Experimental Watersheds of east Texas.

Erosion from a Denton, Texas natural gas hydraulic fracturing well resulted in the loss of 54,000 kg ha^{-1} yr^{-1} of sediment (Mcbroom et al. 2012). Mcbroom et al. (2012) results found that the runoff from hydraulic fracturing wells contributed to 24.67 cm of runoff to the adjacent surface water in 2009. Water quality degradation of the second stream was observed in the form of high salinity values. The source of high salinity values was possibly due to runoff and erosion near a recoded hydraulic fracturing spill. Both Mcbroom et al. (2012) and Burton et al. (2014) concluded that paved and compacted dirt roads provided a pathway for increased runoff and sedimentation. Mcbroom et al. (2012) found in their case studies that silt fences can be ineffective when preventing erosion. This is usually due to improper installation and overwhelming soil volumes that exceed the fences design capacity. Burton et al. (2014), Entrekin et al. (2011), and Mcbroom et al. (2012) all confirmed the increase for potential erosion, runoff, and sedimentation with the clearing and removal of vegetation in the well pad area. Observations from Mcbroom et al. (2012) found that riparian zones were significantly more effective than silt fences.

The environmental concern that receives the most attention among stakeholder regarding environmental impacts of hydraulic fracturing operations is potential groundwater contamination. Since hydraulic fracturing operations involve the injection of millions of gallons of fluids below aquifers and into bedrock, it generates speculation regarding the impact on aquifers (Osborn et al. 2011). Aquifers are a source of water supply for almost half of Pennsylvania residents, making aquifer contamination the environmental concern that dominates public debate (PADEP 2002).

Rabinowitz et al. (2014) conducted a health survey for households dependent on aquifer groundwater near natural gas hydraulic fracturing wells in southwestern Pennsylvania. The study found a correlation between residents living in close proximity of active wells and experiencing skin and respiratory irritation. Osborn et al. (2011) conducted a study monitoring methane levels of 68 aquifer wells located in northern Pennsylvania and New York which overlay the Marcellus Shale and Utica Shale formations. The study found that methane levels were 17 times higher on average in shallow aquifers in close proximity of active well sites. Osborn et al. (2011) found methane concentrations in Pennsylvania groundwater underlying active wells exceeded the U.S. Department of Interior's defined action level for hazard mitigation which is 10-28 mg/L.

Jackson et al. (2013) built on the data collected by Osborn et al. (2011) with a total of 81 wells sampled across six counties in Pennsylvania (Wayne, Wyoming, Bradford, Sullivan, Susquehana, and Lackawanna). 82% of the wells monitored in the study had methane concentrations present in the water supply. Wells located in close proximity to active hydraulic fracturing sites on average had methane levels 6 times higher than wells not adjacent to active fracturing sites. Isotopes in the aquifers concluded that the source of the methane and other gases came from the bedrock elevations. This result means that the source of the methane and other gases in aquifers is most likely from leaky wellbore casings. Wellbores leaks can arise from thermal stress, corrosion, or poor threading. In a study of 7 Marcellus Shale and 1 Barnett Shale groundwater wells, Darrah et al. (2014) utilized noble gases to trace the source of isotopes of methane in aquifers. Darrah et al. (2014) results found that the methane and hydrocarbons in the shallow aquifers did not come from newly formed fractures, but faulty wellbore casings. The defective wellbore casings are the likely source leaking methane into the shallow aquifers. This leaking is most likely due to faulty cement encasing the wellbores.

Llewellyn et al. (2015) studied tap water samples taken from several households in Pennsylvania. The residences were selected based on their tap water connections to groundwater aquifers underlying hydraulic fracturing wells that experienced foaming and inundation. 2D chromatography and mass spectrometry were applied to analyze the samples in a lab. The results found concentrations of the proppant chemical 2-n-butoxyethanol at levels of nanogram-per-litre concentrations which is known to have carcinogenic impacts (Ferrar et al. 2013). The cause of these concentrations is most likely due to leaking wellbores (Llewellyn et al. 2015).

One recent concern regarding hydraulic fracturing is the correlation with seismic and earth activity disturbances (USGS 2015). Geological risks correlated with hydraulic fracturing have been present in communities with active wells. The pressurized water from the injection of proppant can alter the potential for exiting fractures to open due to changes in the in situ stress field (USGS 2015).

Earthquakes have increased in areas experiencing an expansion in hydraulic fracking practices (Healy 2012). The Marcellus Shale Formation of the Appalachian Basin of Pennsylvania has low levels of seismic activity. Despite thousands of wells having been drilled in Pennsylvania since 2005, only six earthquakes have been larger than magnitude 2 (Ellsworth 2013). A lack of earthquakes in Pennsylvania results in Marcellus shale stakeholders concerns towards seismic activity to remain very low.

2.4 Chapter Summary

This chapter described the hydraulic fracturing process and its impacts on water sources within Pennsylvania. The use of AMD provides potential relief for water sources through saving uncontaminated water and removing AMD from surface waters in Pennsylvania. The properties and impacts of AMD are discussed in Chapter 3.

3.0-Acid Mine Drainage Impact on Pennsylvania Water Sources

Pennsylvania produced 25% of the domestic output of coal in the United States over the past 150 years (USGS 2015). Currently Pennsylvania is the fourth leading coal producing state in the nation. The coal deposits of eastern Pennsylvania are classified as anthracite while the deposits that underlie the western region of the state are bituminous coal (USGS 2015). Anthracite coal is used to heat homes and generate electricity while bituminous coal is primarily used for electricity generation. Coalfields are located within the four major river basins in the state which are the Delaware, Ohio, Potomac, and Susquehanna River basins (USGS 2015).

The dominant water quality problem for all four river basins in Pennsylvania is the drainage of abandoned mines into more than 3,000 miles of streams and adjacent groundwater bodies. High concentrations of sediment, acidity, and metals degrade fish habitat, sometimes resulting in streams with no fish (USGS 2015). The current impacts of AMD on Pennsylvania waterways generate an annual loss of $67 million in lost revenue from recreational fishing. Estimations of restoring AMD impacted watershed range from $5 to $15 billion. Active mines must neutralize mine water to a pH of 6-9 before discharging into waterways. Approximately half of the AMD discharges from underground and surface mines are acidic. The most common form of treatment for acid mine discharges is limestone and other calcareous strata (USGS 2015).

The use of the Surface Mine Conservation Reclamation Act (SMCRA) can act as a gateway for addressing the adverse effects of AMD. There are risks to relying on SMRCA as a source of funding and power which are the complications of fund distribution. It will take 50 years for West Virginia to complete their planned remediation projects at the rate they receive funding from the Abandoned Mine Reclamation Fund (Beck 2004). One finding Wood (1996) concluded is that water quality is naturally improving over time since active coal mining had severely decreased in Pennsylvania. Even if there are few results from projects related to limestone treatment and SMCRA it is crucial that active coal mining and generation of AMD remain on the decline and environmental monitoring on the rise. There are no obvious answers to eliminating AMD, but limestone treatment and restoration projects funded by the SMCRA accompanied by environmental monitoring has kept Pennsylvania on the path to recovery from AMD impacts (Beck 2004).

3.1 Acid Mine Drainage Regulations and Liability

In 1976 the federal government enacted the Resource Conservation and Recovery Act (RCRA) that amended the Solid Waste Disposal Act of 1965. One of the most significant amendments in RCRA is Subtitle C which allows the federal government to regulate the disposal of hazardous waste from the point source of generation to the ultimate point of disposal (Luther 2013). This clause applies to waste generators, transporters, operators of waste disposal facilities, and storage facilities. In the Preamble of Subtitle C of RCRA, the EPA indicated that it would prefer to not apply the mentioned regulations on "special wastes". These wastes include, cement kiln dust, utility waste, fly ash, bottom ash, scrubber sludge, phosphate mining and processing waste, uranium and other mining waste, oil drilling muds, and oil production brines.

The reason for the EPA wanting to exclude these hazardous wastes is because "… it occurs in very large volumes, that the potential hazards posed by the waste are relatively low, and that the waste generally is not amenable to the control techniques developed by EPA." (Luther 2013) This exclusion allowed for the Solid Waste Disposal Act of 1980 proposed by Senator Lloyd Bentsen and Representative Thomas Bevill. Commonly referred to as the Bevill amendment, this act excludes the regulation of the "special wastes" as long as it did not interfere with the Safe Drinking Water Act (SDWA) or if states wanted to regulate the special wastes further. Discharges into surface water can be granted through permits from the National Pollutant Discharge Elimination System (NPDES). AMD is considered a special waste under the Solid Waste Disposal Act of 1980 which has resulted in the continued presence of AMD across the Appalachian region through permits from the NPDES that allow for its presence and discharge (Luther 2013).

3.2 Acid Mine Drainage Water Quality Parameters

Acid mine drainage (AMD) is classified as the elevated concentrations of sulfate and dissolved iron particulate through the oxidation of pyrite.

$FeS2 + 3.5\ O2 + H20 \Rightarrow Fe2+ + 2\ SO42- + 2\ H+$ (1)

$Fe2+ + 0.25\ O2 + 2.5\ H2O \Rightarrow Fe(OH)3(s) + 2\ H+$(2)

Half of the acid is the result of oxidation of pyritic iron (the second reaction) while the other half is produced by oxidation of pyritic sulfur (Cravotta et al.1999). Mines that produce AMD have underground voids and sediments with rubble and rejected coal aboveground. The materials above ground are where the pyrite is oxidized. These reactions lead to the common symptoms of AMD which are acidity, toxic metal concentrations, salinization, and sedimentation. The conditions created by AMD leave many streams in eastern Pennsylvania lacking organisms (Cravotta et al. 1999).

There are multiple factors affecting the acidification of streams, with a variety of symptoms that make each water body unique with respect to AMD. Mineralogy of the host rock in streams is a factor when comparing water quality to different watersheds with AMD. The quantity of water in a stream and the path which the water travels affect water quality. Volume and path can impact the amount of dissolved oxygen, which is important during the oxidation of pyrite. The mining method used in the area creates differences among streams that have AMD. Generally, underground coal mines discharge higher concentrations of sulfate and iron than strip mines (Wood 1996). Many abandoned mines have pumps and diversion systems that have not been maintained and facilitate AMD. Surface exposure to sulfide, which facilitates the AMD process, along with variations in exposure time also leads to differences in AMD among sites. Residence time of water in mines is one of the most important variables when comparing AMD impact on streams. Path flow and mine circulation depth are interconnected with residence time because they all relate to exposure of mine waste and the varied sources of AMD (Wood 1996).

Heavy metals and acidity are the dominant hazards in AMD necessary for removal in order to meet EPA standards for domestic water use and aquatic organisms. Heavy metals that often exceeded the EPA standards in AMD are nickel (Ni), cobalt (Co), zinc (Zn), iron (Fe), aluminum (Al), sulfate (SO4), and beryllium (Be) (Cravotta 2007). Heavy metals are of concern because of the detrimental health impacts when taken up by humans, vegetation, and wildlife.

Heavy metals can be adsorbed in soils and spread through waterways where they can bioaccumulate within organisms (Akoto et al. 2014). Even at small concentrations heavy metals can be harmful to humans since they tend to attack specific organs (Naser et al. 2011) and regulate the human metabolic system (Lokeshappa et al. 2012).

Heavy metal transfers from soil to plants to humans are common routes of exposure (Jolly 2013). The vegetation removes heavy metals from soils through consumption (Costello 2003). A common method for removing heavy metals in streams impacted by AMD is through the engineering of wetlands (Cravotta 2010). Leafy vegetables have an affinity for heavy metals, compared to stems and fruits. These metals can impact the development of fish and crops which are then passed on to animals higher up on the food chain when taken up by humans, vegetation, and wildlife (Naser et al. 2011).

Removal of heavy metals from soils and groundwater is very expensive and time consuming. This difficulty in removal is primarily due to heavy metals not biodegrading, being reactive and being easily transported through environments. Nickel and zinc for example tend to adsorb to clays and other heavy metals such as iron and magnesium. Techniques for removing these contaminants often involve bioremediation (Mclean and Bledsoe 2012).

3.3 Pennsylvania Acid Mine Drainage Water Quality-Case Study

In a statewide USGS study of AMD water quality, Charles A. Cravotta (2007) sampled 140 sites including both anthracite and bituminous AMD in Pennsylvania. The study built on data from the same sites used in previous USGS studies from 1985, 1996, and a 1998 study by the Southern Allegheny Conservancy. Samples were taken during the summer and fall of 1999 during base flow conditions. The study summarizes the pH and constituents mostly in concentrations of trace metals and rare earth metals (Cravotta 2007).

Sulfur in the form of SO4 was the most dominant nonmetal in the study of all 140 sites. 84% of the samples exceeded the 240 mg/L EPA drinking water standards for secondary contaminant level with a median concentration of 520 mg/L (Cravotta 2007). The median SO4 concentration is much higher than the 10.7 mg/L global river water average, but significantly less than the average seawater concentration of 2715 mg/L (Cravotta 2007). 84% of AMD samples had Se concentrations higher than the 0.2 µg/L detection level and close to the average Se concentrations for seawater. No samples exceeded the maximum contamination level of 50 µg/L for Se. Carbon (C), nitrogen (N), oxygen (O), and phosphorus (P) had moderate concentrations when compared to river water and seawater averages (Cravotta 2007).

The dominant halogen in the samples was chloride with a median concentration of 7.3 mg/L. 90% of the samples had concentrations of I and Bromide greater than 0.003 mg/L (Cravotta 2007). All sample concentrations of Cl and Br were significantly lower than seawater concentration averages (Cravotta 2007). Na and potassium (K) were the dominant alkali cations while lithium (Li), rubidium (Rb), and cesium (Cs) in the samples has much lower concentrations. Na, K, and Rb had concentrations similar to average river water and much less than seawater concentrations. Li, Rb, and Cs median concentrations exceeded those of river water concentrations while only the top 5% concentrations exceeded seawater concentration averages (Cravotta 2007).

Calcium and Mg had concentration median of 88 and 38 mg/L making them the dominant alkaline earth metals of the AMD samples (Cravotta 2007). Ca, Mg, and Be concentration medians were elevated compared to average river water, but significantly lower than seawater concentration averages. Ba concentrations were comparable to river and seawater averages falling below MCL concentration levels. Be concentrations exceeded river water and seawater by two or three times and 1/3 of the samples exceeded the Be drinking water MCL of 4 µg/L (Cravotta 2007).

Silicon had the highest concentration levels of metalloids with a median level of 7.9 mg/L (Cravotta 2007). Median concentrations for B were 44 µg/L, As 1.7 µg/L, Ge 0.07 µg/L, and 0.01 µg/L for antimony (Sb). The silicon median concentration exceeded that of seawater, but not river water. B and Ge median concentrations exceeded levels in river water, but were less than or equal to that of seawater (Cravotta 2007). As and Sb had median concentrations less than river water except for bituminous concentrations for As which exceeded river water median concentrations. Sb Ge, and As concentrations fell below CCC levels. 10% of samples had As concentrations that exceeded drinking water MCL levels of 10 µg/L (Cravotta 2007).

The transition metals in order of abundance Fe, Mn, Zn, Ni, Co, Ti, Cu, and Cr had median concentrations that exceeded criteria for river water, seawater and aquatic life (Cravotta 2007). 80% of the samples taken exceeded the Fe PME criteria of 7.0 mg/L and the Mg criteria of 5.0 mg/L. 95% of the samples exceeded the drinking water SCL values for Mn of 0.05 mg/L and Fe of 0.3 mg/L. Most of the samples exceeded the CCC freshwater values for multiple transition metals. Aluminum was the second most abundant metal behind Fe (Cravotta 2007).

None of the 140 AMD sites sampled met the EPA criteria for the protection of aquatic organisms due to the elevated concentrations of As, Al, Be, Mn, Fe, SO4, and Zn (Cravotta 2007). 33% exceeded primary drinking water standards for Be and 10% exceeded those for As. 139 samples failed to meet drinking water standards for Al, Fe, Mn, and SO4. 137 samples failed for mine-effluent criteria for Al, Fe, Mn, SO4, pH, and net acidity. Bituminous AMD samples had higher concentrations of Al, As, B, Cu, Fe, Mn, Ni, Se, SO4, alkalinity, and acidity. Anthracite coal had higher concentrations of Pb and Ba (Cravotta 2007).

3.4 Pennsylvania Bituminous Acid Mine Drainage Remediation Case Studies

Charles A Cravotta et al. (1998) conducted a treatment study on a 66 hectare reclaimed bituminous coal site on a hilltop in Clearfield County, Pennsylvania. The altitudes of the site ranged from 540m to 580m. The study area overlays a formation of shale, underclay, and silt. Groundwater networks in the hillsides transport the contamination of AMD into local surface waters. The coal range the site resides in is the Kittanning coal range which was sporadically mined from 1965 to 1985 (Cravotta et al. 1998). Mining operations ended in 1985 but the pits remained open until reclamation efforts began which involved revegetating, regrading, and backfilling completed in 1988. Revegetation proved difficult due to dry and acidic soils. A combination of wood chips and sewage sludge was applied to 70% of the site at a rate of 134.5 Mg/ha (megagrams per hectare). The sludge contains a mixture of 35% woodchips, 20% sludge solids, and 45% water (Cravotta et al. 1998). The pH of the mixture is 5.5 with 0.3% potassium, 1.5% phosphorous, and 2.3% nitrogen. Limestone powder was also applied to the site near the root zone. The sludge also contained enriched amounts of cadmium, chromium, copper, nitrogen, phosphorus, nickel, lead, and zinc. Compared to elevated concentrations of metals and nutrients from the mine runoff, the sludge added negligible amounts (Cravotta et al.1998).

From 1989 to 1990 a network of 7 wells were monitored after the application of sludge to the surface soil (Cravotta et al. 1998). 3 of the 7 wells were treated with sewage while 3 were untreated. One was an untreated and unmined well acting as a control. The unmined well had alkalinity levels of 53 to 140 mg/L CaCO3 alkalinity and a median pH of 6.6 (Cravotta 2008). The concentrations of sulfate were less than 130 mg/L, nitrate concentrations were 0.2 mg/L, and iron concentrations were 4.5 mg/L (Cravotta et al. 1998). Levels of sulfate, iron, and pH for untreated wells were 410 mg/L, 0.9 mg/L, and 4.4. Treated wells had sulfate, iron, and pH levels of 260 mg/L, 0.6 mg/L, and 5.9. The organic (carbon), inorganic (nitrogen, nitrate, and sulfate), and metals (aluminum, cadmium, chromium, nickel, manganese, and zinc) were elevated in groundwater wells that had been treated with sludge (Cravotta et al. 1998).

The sludge application did increase the amount of vegetation cover on the surface of the reclaimed mine site (Cravotta et al. 1998). However the sludge did not act as an effective barrier to the consumption of O2 failing to prevent the oxidation of pyrite and increased acidification of the groundwater. Sludge treatment was also demonstrated to further degrade groundwater through elevated concentrations of metals and nutrients. The treatment proved to be effective with the increase of groundwater pH through the limestone (CaCO3) present in the sludge. Biodegradation of the sludge is rapid so the impacts of the treatment will be short lived (Cravotta et al. 1998).

Charles A. Cravotta (2005) conducted a study of the Staple Bend Tunnel Unit of Allegheny Portage Railroad National Historic Site (ALPO-SBTU) test site in Cambria County, Pennsylvania. The purpose of the study was to measure the quantity and quality of the AMD present at the site in order to assess what remediation methods could be applied to the AMD impacted site. In April 2004, samples of AMD were collected from 8 sites in the tunnel diverting the Conemaugh River water and an adjacent pond. Steel lag and limestone from local sources were applied to AMD samples in the lab for analysis to compare the two different treatment methods (Cravotta 2005).

The water quality of the AMD samples from the upper pond near the outflow had concentrations of dissolved manganese of 0.84 mg/L and 0.91 mg/L, iron of 0.25 mg/L and 0.41 mg/L, aluminum of 5.07 mg/L and 3.94 mg/L, silica of 15.7 mg/L, and sulfate of 356 and 358 mg/L, pH of 3.8 and 3.5, and net acidity of 38 and 41 mg/L as CaCO3 (Cravotta 2005). The upper pond sampling site near the inflow source had flow rates of 260 and 324 gal/min with dissolved manganese of 1.48 and 1.46 mg/L, iron of 0.45 and 0.60 mg/L, aluminum of 9.25 and 7.62 mg/L, silica of 22.7 and 24.0 mg/L, and sulfate of 536 and 499 mg/L. The combined data for the other sites had a flow of 955 gal/min with dissolved manganese of 0.93 mg/L, iron of 0.35 mg/L, aluminum of 3.87 mg/L, silica of 15.8 mg/L, and sulfate of 706 mg/L with pH of 3.7 and net acidity of 32 mg/L of as CaCO3 (Cravotta 2005).

Treatment methods were conducted in the lab with application of limestone or steel slag to AMD from two sites with different water quality conditions (Cravotta 2005). One site had DO levels of 11.2 mg/L, dissolved aluminum of 4.96 mg/L, iron of 0.96 mg/L, and manganese of 1.48 mg/L. The other site had lower dissolved oxygen and higher concentrations of metals with DO of 0.7 mg/L, dissolved aluminum of 9.31 mg/L, iron of 71.3 mg/L, and manganese of 4.40 mg/L. Both treatments to both sites increased pH, alkalinity, and calcium concentrations. Both treatment methods decreased the concentrations of aluminum and iron while limestone increased manganese and steel slag increased silica. Except for barium and strontium limestone and steel slag overall decreased the concentration of dissolved trace metals. Limestone had the most impactful results with improving water quality as a long-term performance (Cravotta 2005).

3.5 Pennsylvania Anthracite Acid Mine Drainage Remediation Case Studies

The anthracite region of Pennsylvania contains four fields that lie within the Ridge Valley Physiographic Province which are underlain by, conglomerate, sandstone, and shale with several coal seams (Wood 1996). Coal underlies the center valleys and in some instances even the ridges. From 1830 to 1972, large amounts of coal were extracted from underground mines within the anthracite regions of eastern Pennsylvania (Wood 1996). Anthracite coal was discovered in the region around 1750, but it did not become profitable to extract until the early 1800's. The annual extraction of coal in the region reached 80 million tons by 1913 and peaked in 1917 at 100 million tons (Schuylkill Conservation District 2005). Rivers in eastern Pennsylvania provided transportation for the coal to international markets.

Anthracite coal mining remained unregulated until the late 1970's, at which time most of the heavy extraction had already taken place. The Federal Water Pollution Control Act of 1972 began addressing pollution issues of the region, followed by the Federal Surface Mining Control and Reclamation Act (SMCRA) of 1977 (Schuylkill Conservation District 2005). Since the decline of coal mining in the region, mines have been left abandoned with visible environmental damage in the surrounding areas. Mine waste piles create small hills that mix with soil and surface mining material left behind results in sparsely vegetated depressions. Water has filled the deep voids from underground mines that reside below the water table. The conditions left behind from the coal mining era have resulted in abandoned mine discharge or acid mine discharge (AMD) which continues to be a dominant factor of water degradation in eastern Pennsylvania (Wood 1996). Schuylkill County has over 108 discharge sites identified by the EPA (Schuylkill Conservation District 2005).

Wood (1996) studied multiple aspects of water quality in streams impacted by AMD in the anthracite coal region of Pennsylvania between 1975 and 1996. He found that water temperatures were higher in areas that had mines with greater depth. This characteristic is related to the geothermal gradient of about 1°C per 100 ft. Strip mines did not have as high of temperatures due to their lack of depth compared to underground mines. The results of a temperature study are not entirely conclusive due to the proximity of urbanized areas, which could be responsible for some increase in temperature (Wood 1996).

Wood (1996) studied 81 sites of AMD pH between 1975 and 1991 and found that 64 of them increased in pH over the study period with a median of +0.4 units. Thirteen of the sites decreased in pH from 1975 to 1991, while 4 of the sites had no change. The pH was measured by previous teams between 1961to 1969 at 23 mines which showed a median increase of +0.8 units at all of the mines (Wood 1996). Measuring direct discharge from mines is difficult because pH can vary due to exposure of the atmosphere. Lab results often have higher pH levels than field results due to hydrolysis and oxidation from atmospheric exposure during sample transportation (Wood 1996).

Acidity measurements by Wood (1996) on the 81 sites do not show an overall increase. Forty-one of the sample sites experienced an increase median of +2 mg/L calcium carbonate. The other 40 sample sites experienced concentration fluctuations over the testing years with some decrease. Large changes in acidity have taken place at specific sites overtime, but not as a whole in the anthracite coal region (Wood 1996). A negative correlation between dissolved oxygen and acidity was found during sampling. Out of the 81 sites, all of them exhibited a decrease in acidity when the dissolved oxygen increased. This was the only relationship found between dissolved oxygen and other water quality indicators. Dissolved oxygen had not been studied previously, making it difficult to draw conclusions relating to AMD in the area (Wood 1996).

The dominant cations in anthracite AMD are calcium, magnesium, iron, sodium, manganese, aluminum, and potassium (Wood 1996). Other trace metals found in anthracite AMD are strontium, zinc, nickel, cobalt, lithium, born, copper, lead, and cadmium. Aluminum concentrations have been measured in 29 mines in the region with very little evidence of a positive trend. Only 4 out of the ten most studied mines for aluminum showed an increase while 6 had decreased concentrations. Groundwater samples outside of the mined area are also quite low, showing inconclusive data with regard to increased aluminum concentrations (Wood 1996).

Barium concentrations measured at 28 mines between 1965 and 1990 were low as expected (Wood 1996). This low value is an indicator of high concentrations of sulfate which creates barium sulfate, an insoluble compound, which leads to low concentrations of barium. Calcium concentrations are usually elevated in AMD samples. A decrease was seen between 1975 and 1991 at sample sites along with magnesium concentrations due to the cation anion balance. Elevated concentrations in cobalt and nickel had been reported in 1971in the Black Creek watershed which could be due to cobalt acting as a sulfide in coal. Cobalt can replace parts of the iron in pyrite, but increases in cobalt had been detected between 1969 and 1990 (Wood 1996).

All of the iron concentrations except for 8 of the 82 discharge sited exceeded the EPA's secondary maximum contaminant level of 0.3mg/L (Wood 1996). Iron is a dominant metal present in AMD due to its affiliation with iron sulfide. Despite the exceeding of EPA standards, 82 of the 85 mines experienced a decrease in the iron concentrations. Some decreases were as high as -100mg/L (Wood 1996). Similar to iron, manganese concentrations exceeded EPA secondary maximum contaminant levels of 0.5mg/L at all sites except two. Between 1970 and1990 there was an overall decrease in manganese concentrations in 23 of 27 mines tested (Wood 1996).

Sodium and chloride are often found in elevated concentrations in the Northern Anthracite Field (Wood 1996). This could be due to urbanization or because of the historical use of NaOH by mines to neutralize AMD. Sludge used by the mine treatment plants in the region dumped NaOH sludge into abandoned mines which could be seeping into the water supply. Nitrite and nitrate concentrations remained low at around 0.1mg/L. It is possible that some of the nitrite and nitrate has been reduced to ammonia. No ammonia testing has been done in the area so nothing can be concluded (Wood 1996).

Since 1960, 65 of the 85 mines tested for sulfate have decreased while 15 mines experienced an increase (Wood 1996). This progress is attributed to the mines closing in the region. Concentrations are expected to decrease into the future, but not as rapidly. Concentrations of lithium, lead, zinc, strontium and potassium were sampled but either due to low concentrations or a lack of historical data nothing conclusive could be found regarding these metals (Wood 1996).

The data gathered by Charles Wood (1996) is one of the first long term studies regarding AMD in the Anthracite Coal region of Pennsylvania. His findings show that there are still deficiencies in water quality in the region. Most of these impairments are in the form of pH, temperature, iron, and sulfate concentrations. Overall the quality of the water in the region has been improving with all of the impairments being minimized since most of the mines had been closed in the 1960's (Wood 1996). The data gathered by Wood (1996) over the past thirty years can act as guidance on what forms of treatment can be used and which water quality impairments should receive the greatest attention.

Multiple treatment methods have been developed to treat watersheds negatively impacted by AMD. State and local agencies developed passive and semi-passive treatments to reduce acidity in water quality and to prevent transporting dissolved metals (Cravotta 2010). Depending on available space, the treatment systems are constructed immediately below the AMD source. Treatment systems in Pennsylvania can be installed by watershed associations and monitored by the Pennsylvania Department of Environmental Protection (PADEP), U.S. Department of Energy (USDOE), and U.S. Geological Survey (USGS) (Cravotta 2010).

Open limestone channels and limestone sand dosing are simple forms of passive treatments. Limestone is added infrequently near the source of the AMD for the purpose of adding alkalinity to acidic streams that may also have high concentrations of aluminum and iron (Cravotta 2010). Trucks can dump several tons of limestone sand directly into the stream which usually will take 5 minutes to dissolve (if the sand is less than 0.5cm in diameter) (Cravotta 2010). Limestone channels can be constructed using ten times as much limestone in the forms of sand and cobbles that range between 3 and 11 cm. The sand and cobbles are dispersed throughout stream beds in berm formations (Cravotta 2010).

Anoxic limestone drains (ALDs) are another simple passive treatment where cobble-sized stones are buried in trenches near AMD contaminated groundwater (Cravotta 2010). When the groundwater and buried limestone come in contact with one another the pH increases before it emerges into the stream. ALDs are the preferred form of treatment over open passive systems because of greater alkalinity generated (Cravotta 2010). Keeping carbon dioxide in the limestone beds increases alkalinity generation and limestone dissolution in water. Compost can be applied to limestone beds for the retention of carbon dioxide. Keeping oxygen out of limestone beds decreases the chance of iron oxidation which accelerates AMD production and can cause clogging in the limestone bed. Allowing for oxygen in the limestone bed has benefits such as the removal of iron, magnesium, and other trace metals that can clog the limestone bed. Pipes can be installed into the limestone beds to discharge the buildup of aluminum and iron oxyhydroxides (Cravotta 2010).

Limestone diversion wells treat AMD by redirecting AMD natural flow into a 1.2 meter pipe that has limestone aggregate inside (Cravotta 2010). The diversion wells churn water while crushing limestone to facilitate dissolution and prevent oxyhydroxides from encrusting. One possibility of the process is that oxyhydroxides could precipitate and accumulate downstream of the well. Limestone diversion wells can consume up to one ton of limestone per week, which requires heavy maintenance (Cravotta 2010).

Settling ponds and constructed wetlands are common AMD treatments. Ponds and wetlands promote metal precipitation and deposition. Anaerobic ponds are used to treat acidic water and aerobic ponds are used to treat alkaline water. If the wetlands or ponds have limestone present in their foundation it can accelerate the treatment process. Microbial activity in wetlands can also reduce the sulfate concentrations (Cravotta 2010).

Depending on specific concentrations of metals, pH levels, and available resources, different forms of treatment may be preferable and will have different results. Cravotta et al. (2010) applied all the treatments to different sections of the Swatara Creek in Schuylkill County, Pennsylvania to see which would be the most effective and efficient for treating AMD in the anthracite coal region of Pennsylvania. For limestone-sand dosing Cravotta et al. (2010) found that calcium carbonate concentrations decreased at Coal Run where the treatment was applied. Concentrations fell from 11.2 mg/L to 1.2 mg/L of calcium carbonate and pH increased from 5.6 to 6.9 making it the most effective of the applied treatments. The open limestone channel treatment applied on Swatara Creek overall decreased the concentrations of metals generated from AMD and decreased the acidity of the water. One concern is that the open limestone channel will become less efficient as the quality of water improves (Cravotta et al. 2010).

The anoxic limestone drain (ALD) installed on Buck Mountain had mixed results as to how effective it was on reversing the impacts of AMD (Cravotta et al. 2010). Overall the ALD treatment increased the pH of the stream and was able to neutralize dissolved metals from the AMD. The ALD was not effective for decreasing concentrations of metals overall and did not function very well when heavy tropical storms occurred in the summer. Similar dysfunctions occurred with the limestone diversion wells installed in Swatara Creek. The wells increased pH levels but during heavy seasonal flows much of the AMD bypassed the wells going untreated.

Anaerobic wetland treatment was successful in removing iron and aluminum concentrations while decreasing pH (Cravotta et al. 2010). An adverse effect of the wetlands is that the water temperature varies significantly, making it unsustainable for fish. Limestone-compost- based wetland treatment was the most expensive and the least effective for removing acid. There was success for the removal of metals but the rate of reduction was not fast enough to prevent metals from being transported during storms (Cravotta et al. 2010).

Bott et al. (2012) in a recent study found that the remediation efforts on the Swatara to not be functioning. Bott et al. (2012) observed that pH fell below 5 in many samples with only positive impact being a decrease in sulfate concentrations. The lack of results from AMD treatment benefiting macroinvertebrate population has furthered complications with stream respiration (Bott et al. 2012). One species that has recovered after AMD treatment is the fish population in the Swatara Creek. From 1996-2006 there were only 6 different species of fish and now there are 25 (Cravotta et al. 2010). This increase could be attributed to the overall increase of alkalinity in the stream, but it will need to be sustained in order for the fish populations to continue to thrive (Cravotta et al. 2010).

The most important leverage point available to Schuylkill County is the Surface Mining Control and Reclamation Act (SMCRA). The SMCRA provides potential money and power to the county which is an ideal recipient of such resources (Beck 2004). Authority from the SMCRA could allow and fund Schuylkill County to build the limestone sand dosing active treatment facilities and infrastructure that would improve water quality. The SMRCA can also provide funding for the purchasing of abandoned mines that can be sealed off which would decrease sources of AMD throughout the county (Schuylkill County Conservation District 2005). Purchasing abandoned or damaged land could also expand pervious soil which decreases erosion and sedimentation. Education about projects and potential projects relating to SMCRA will be crucial if the Department of Agriculture is to form partnerships with land and water rights owners that have experienced loss or damage from AMD. Partnerships between private individuals and the government agencies can be a source of financial relief for the county and residents, while improving public health and safety (Schuylkill County Conservation District 2005).

3.6 Chapter Summary

This chapter describes the environmental impacts of acid mine drainage (AMD) on water sources in Pennsylvania. The water quality properties of both bituminous and anthracite AMD are compared and contrasted in the first case study. Bituminous and anthracite AMD remediation case studies are analyzed in the last two sections to explain the costly processes that yield limited results. Chapter 4 examines utilizing AMD in hydraulic fracturing operations in Pennsylvania.

4.0-Utilizing Acid Mine Drainage for Hydraulic Fracturing in Pennsylvania

Treating hydraulic fracturing flowback fluids (HFFFs) is an emerging concern for government entities and stakeholders impacted by the fracking process. Certain constituents such as radium and anions bring into question whether HFFFs can be treated using conventional wastewater facilities. A recent study by Kondash et al. (2013) has shown lab results that when HFFs are blended with acid mine drainage (AMD) there are sharp decreases in sulfite, iron, barium strontium and radium. Watershed groups in Pennsylvania have been encouraging the use of AMD in fracking to bring relief to impacted watersheds. This method could potentially treat HFFFs, while saving uncontaminated water, and remediating streams (PADEP 2014).

The Pennsylvania Department of Environmental Protection (PADEP) White Paper: Utilization of Mine Impacted Waters for Natural Resources Extraction Activities proposes to extend the Environmental Good Samaritan Act to fracking operations. This extension would allow for storage of AMD on fracking sites in the form of pools, tanks, and pits. The PADEP has stated recommended concentrations for heavy metals only for storage, but not for transport or treatment activities (PADEP 2011). This will require the transporting of AMD to storage sites in the Marcellus Shale region. Storage of AMD to possibly hundreds of wells in Pennsylvania could potentially increase the geographic area contaminated by AMD.

Dozens of unconventional hydraulic fracturing wells are located in watersheds that contain high quality Chapter 93 streams. Streams classified as high value or high quality exceed EPA water quality standards which provide sustainable ecological habitat for fish and wildlife. Chapter 93 streams are protected and maintained under the Pennsylvania Commonwealth Code under water quality standards (PADEP code Chapter 23). The 83,000 miles of Pennsylvania streams provide water for multiple industries that provide employment and tax revenue for state residents (U.S. Fish and Wildlife 2015).

4.1 Mixing Acid Mine Drainage and Hydraulic Fracturing Flowback Fluids - Case Study

Kondash et al. (2013) conducted a study in which AMD and HFFFs were blended in a lab. The purpose of the study was to see if specific constituents of concern could be sequestered or diminished when the two forms of waste were blended together. The study focused on the levels of naturally occurring radioactive material (NORM) and heavy metals (Kondash et al. 2013). Lime treated AMD samples were gathered from AMD sites in western Pennsylvania while synthetic AMD was generated to represent iron rich untreated AMD sources. Lime treated AMD had a pH of 10-11 while the synthetic untreated AMD had a pH of 3.5. HFFF samples were sampled from three hydraulic fracturing sites within close proximity of the treated AMD sites sampled. Six sets of treated and untreated AMD were mixed with the HFF samples using 25%, 50%, and 75% AMD (Kondash et al. 2013).

Lab results showed that all mixtures of AMD and HFFF resulted in the reduction of Ba, Sr, and Ra for HFFFs and the removal of SO4 and Fe for treated AMD (Kondash et al. 2013). The removal of Ba, Ra, and Sr increased with higher concentrations of AMD in the mixture. Treated AMD removed more Ba, Ra, and Sr than the untreated AMD. Sulfate was removed from all samples as well, but the higher the concentration of AMD in the mixture the less sulfate was removed. Fe removal was observed in all mixture variations with higher removals with untreated AMD. 100% of sulfate, Ba, and Ra were achieved during the different trials while 75% of Sr and 97% of Fe were removed (Kondash et al. 2013). To achieve the highest removal of all the constituents it would be best to use a mixture with high fractions of lime treated AMD.

The removal of hazardous constituents is correlated with the oversaturation of several minerals depending on the type of AMD and the mixture fractions (Kondash et al. 2013). Saturation of barite and celestite were found to increase with the increasing concentration of AMD and sulfate in the mixtures. Calcite saturation was positively correlated with pH. Lower pH AMD with higher Fe concentrations resulted in higher iron-bearing mineralization such as siderite and hermatite. The dominant precipitants for treated and untreated AMD were calcite, celestite, and barite. Other minerals found in mixtures were strontium, barite, and quartz. Presence of these minerals and precipitants from mixing AMD and HFFFs could possibly cause scaling on the walls of wellbores during hydraulic fracturing operations and require routine maintenance (Kondash et al. 2013).

4.2 Utilization of Acid Mine Drainage for a Fracking Operation - Case Study

ProChemTech International Inc. (2009) conducted the first study and application of treated AMD for a hydraulic fracturing operation near Hawk Run, PA. The study involved the extraction of AMD from the Blue Valley Fish Culture Station (BVFCS) and treated for utilization in a fracking operation in Hawk Run, Pennsylvania.

Treated and untreated samples of AMD from BVFCS were collected for the study to compare with AMD mixed with HFFFs (ProChemTech International Inc. 2009). The untreated AMD had concentrations of Ba <0.2 mg/L, Ca 196 mg/L, Fe 13.0 mg/L, Mg 56.0 mg/L, Mn 56.0 mg/L, Sr 3.6 mg/L, CacO3 752.7 mg/L, and TDS 1,004 mg/L. The treated AMD had concentrations of Ba <0.2 mg/L, Ca 198 mg/L, Fe 0.32 mg/L, Mg 55.5 mg/L, Mn 5.54 mg/L, Sr 3.6 mg/L, CaCO3 734.5 mg/L, and TDS 1,076 mg/L (ProChemTech International Inc. 2009)

When the treated and untreated AMD were mixed with HFFFs in the lab most concentrations among the constituents were significantly impacted (ProChemTech International Inc. 2009). The untreated AMD had concentrations of Al 3.4 mg/L, Ba <0.1 mg/L, Ca 154 mg/L, Fe 58.5 mg/L, Mg 65.5 mg/L, Mn 5.45 mg/L, Sr 0.12 mg/L, CaCO3 788.6 mg/L, TDS 1,004 mg/L. The treated AMD had concentrations of Al <0.1 mg/L, Ba <0.1 mg/L, Ca 16 mg/L, Fe <0.03 mg/L, Mg 42.0 mg/L, Mn <0.04 mg/L, Sr <0.02 mg/L, CaCO3 212 mg/L, and TDS 1,520 mg/L.

The concentration levels were acceptable since they were below the concentration objectives of Ca <350mg/L and Fe < 20 mg/L (ProChemTech International Inc. 2009). A primary reason why these concentration were able to be achieved is due to the removal of iron at the BVFCS treatment facility and the absence of calcium hydroxide used in the treatment as well. The cost to treat the AMD at BVFCS was calculated at $2.5/1,000 gallons. The estimated cost of building a treatment facility similar to BVFCS that can treat 720,000 gallons/day operated by two men is $1 million (ProChemTech International Inc. 2009).

4.3 Pennsylvania DEP White Paper Summary

The Pennsylvania DEP has published the White Paper: Utilization of AMD in Development for Natural Gas Development which was written by a staff of DEP staff member to establish a process for natural gas hydraulic fracturing operations to utilize AMD (PADEP 2011). The document identifies AMD sources to be used by hydraulic fracturing operations and how to store AMD on hydraulic fracturing sites. Other topics of concern addressed in the white paper are permitting, liability and coordination between different departments (PADEP 2011).

The storage options for AMD on hydraulic fracturing sites will depend on the water quality of the AMD source (PADEP 2011). AMD will have to meet certain water quality standards in order to obtain storage permits for different facilities. Alkalinity minimum of 20 mg/L, Al 0.2 mg/L, Ammonia 1.0 mg/L, Arsenic 10.0 µg/L, Ba 2.0 mg/L, Br 0.2 mg/L, Cd 5.0 µg/L, Chloride 250 mg/L, Cr 100 µg/L, Cu 1.0 mg/L, Fe mg/L 1.5 mg/L, Pb 1.5 µg/L, Mn 0.2 mg/L, Ni 470 µg/L, pH 6.5-8.5 phenol 5.0 µg/L, SE 50 µg/L, Conductivity 1000 µmho/cm 1,000, Sulfate 250 mg/L, TDS 500 mg/L, TSS 45 mg/L, and Zn 5.0 mg/L (PADEP 2011).

One type of storage that is an option for AMD utilized in hydraulic fracturing operations is a centralized freshwater impoundment which is a facility that stores freshwater for multiple well sites (PADEP 2011). It can be located on or separate from a well location. These types of centralized impoundments are jurisdictional and nonjurisdictional. A jurisdictional impoundment is not located on a watercourse and may not have any drainage into a waterway that has a depth greater than 15 feet. The storage of the impoundment may not exceed that of 16.3 million gallons. Nonjurisdictional impoundments are used for freshwater or semifluids that do not pose a threat to persons or property in the form of pollution or danger. They also may not be on a watercourse or contribute to streams with a depth of 15 feet and a maximum storage of 16.3 million gallons. Nonjurisdictional impoundments may have to obtain additional permitting for erosion and soil disturbances (PADEP 2011).

Centralized wastewater impoundment dams for oil and gas activities store wastewater for servicing multiple wells (PADEP 2011). These impoundments are not to be used for residual waste storage and may be located on or near adjacent well sites. The specific impoundment must have a primary liner, a leak detection zone, a secondary liner with a thickness no less than 40 millimeters, and a groundwater monitoring system. A dam permit is also required before construction of the facility. On site pits and tanks can be used to store wastewater and freshwater for servicing a single well and they must have a primary liner of 30 millimeters thick, but require no leak detection or groundwater monitoring (PADEP 2011).

Options for addressing long term liability for parties utilizing AMD for hydraulic fracturing wells are the applications of the Environmental Good Samaritan Act (EGSA) or the use of a Consent Order and Agreement (PADEP 2011). The EGSA would grant parties using AMD for hydraulic fracturing immunity from civil liability under state law. Parties would include landowners and those who supply equipment or materials at no cost for “water pollution abatement projects”. These projects are defined as treatment of polluted waters on abandoned lands or the treatment of AMD. Immunity of liability for maintaining and operating water abatement facilities would be granted to these parties as well (PADEP 2011).

The Consent Order and Agreement excludes PADEP from saddling parties using AMD for hydraulic fracturing with long term liability so long as specific conditions are met by the operator (PADEP 2011). The goal of PADEP would be to provide treatment for the AMD source after hydraulic fracturing operations had ended. This effort could include selling treated AMD from existing treatment facilities and depositing the revenue into a trust fund used for mining programs. If a treatment facility needs to be constructed by the hydraulic fracturing operation then the operator could sign over the facility to a non-profit organization or government agency after drilling operations have ended. The objective of the Consent Order and Agreement Act is to provide sustainable funding for treatment facilities both during and after hydraulic fracturing operations (PADEP 2011).

4.4 Chapter Summary

This chapter looks at two case studies involving the mixing of AMD with HFFF and the treatment results. The last section summarizes PADEP recommendations for storage and liability of AMD for hydraulic fracturing operations. The final chapter discusses the potential benefits, risks, challenges, conclusions, and recommendations for utilizing AMD in hydraulic fracturing operations.

5.0 - Analysis of Utilizing Acid Mine Drainage for Pennsylvania Hydraulic Fracturing

Hydraulic fracturing is a complex and polarizing issue for Pennsylvania stakeholders, especially with regard to impacts on water sources. AMD continues to degrade over 3,000 miles of streams from over a century of coal mining with limited results from remediation projects. Extracting AMD from waterways could bring needed relief to impacted streams, but it also generates more questions and concerns for an already complex issue. Chapter 2 described the hydraulic fracturing mechanics and environmental impacts on water sources and Chapter 3 looked at AMD water quality parameters along with remediation case studies. The purpose for this approach was to see how AMD could provide environmental relief or increase environmental risks for Pennsylvania water sources. Chapter 4 provides information on whether hydraulic fracturing operations utilizing AMD could assist in remediation of impacted streams. Chapter 5 combines information from the previous chapters to assess benefits, risks, challenges, conclusions and recommendations.

5.1 Potential Benefits for Pennsylvania Water Sources

Utilizing AMD in the hydraulic fracturing process is a practice being proposed by stakeholders to potentially save clean water, remove AMD impacted water, and treat HFFF without treatment plants. The literature cited to analyze this topic shows that these benefits are possible, but do not share the same magnitude. Certain benefits will be more significant than others due to environmental conditions and industrial practices.

The benefit that would have the most impact is saving clean water from being utilized in the hydraulic fracturing operations which leads to contamination and the generation of HFFF. Mantell (2011) says that the volume of water needed to drill a hydraulic fracturing well is estimated at 5.6 million gallons in the Marcellus Shale region. The PADEP issued permits for the establishment of 1,652 unconventional natural gas wells for 2013 (PADEP 2013). If only 1,000 of these wells are drilled with a mixture of 75% AMD with water and proppant, billions of gallons of water could be saved based on the equation:

(5.6 million gallons of water) x (1,000 wells) x (75% AMD) = 4.2 billion gallons of water saved

If the average 10-15% of the mixture flows back (Mantell 2011), then a potential 420 million gallons could be recycled for future drilling operations, which is contingent on the HFFF and AMD mixture treatment having success in the field.

If treatment of HFFF using AMD on site during drilling operations is successful, then this would be the most impactful benefit behind saving clean water. Kondash et al. (2013) and ProChemTech (2009) were able to treat HFFF in labs using treated and untreated AMD. Their treatments were able to remove the same constituents, such as Ba and Ra, better than the treatment plants studied by Warner et al. (2013) and Ferrar et al. (2013). AMD treatment could allow for the reuse of HFFF, preventing residual contaminants such as benzene, Cl, and 2-butoxyethanol from entering surface waters when discharged from treatment plants into rivers.

Removal of AMD from streams through utilization alone does not seem probable without constructing more AMD treatment sites. As seen in the studies conducted by Kondash et al. (2013) and ProChemTech (2009) HFFF showed greater reduction in contaminants with treated AMD rather than untreated AMD. 16 acid mine treatment facilities were constructed in Pennsylvania from 1967 to 1992 costing from $40,000 to $5 million (PADEP 2012). If nonprofits and government agencies partner with hydraulic fracturing operations using the SMRCA, it is possible to increase the number of AMD treatment facilities within Pennsylvania. Utilizing treated AMD from existing AMD treatment facilities for hydraulic fracturing operations would not increase benefit, but potentially further competition between stakeholders.

5.2 Potential Risks for Pennsylvania Water Sources

Utilizing AMD for hydraulic fracturing sites will generate risks though potentially spreading contaminants within AMD to uncontaminated water sources. Studies have shown that surface water and aquifers have been impacted due to hydraulic fracturing operations. If AMD were present and utilized during fracturing operations, contaminants such as heavy metals could be deposited into water sources that have not experienced AMD impacts.

Research from Burton et al. (2014), Entrekin et al. (2011), and Mcbroom et al. (2012) indicated that fracking operations increased runoff, sedimentation, and erosion. If spills were to occur while utilizing AMD on hydraulic fracturing sites, AMD contaminants could adsorb to soil. Adsorption to soil could threaten vegetation and facilitate mobilization into water bodies such as aquifers and surface waters. Osborn et al. (2011), Jackson et al. (2013), and Darrah et al.

(2014) discovered that methane contamination of Pennsylvania aquifers underlying hydraulic fracturing operations was due to leaking wellbores. Garth et al. (2015) discovered the proppant constituent 2-n-butoxyethanol in several tap water samples connected to a Pennsylvania aquifer underlying hydraulic fracturing operations. If treatment of AMD and HFFF is not successful and the mixture is injected into a leaking wellbore, then there is potential for AMD contamination of the aquifer. Kondash et al. (2013) also stated that mineralization from mixing AMD with HFFF could potentially cause scaling and damage to wellbores during mixing, which could potentially increase the leaking of wellbores into aquifers.

HFFF and AMD are exempt from the Clean Water Act of 1973 due to the Bevill Amendment of 1981 and the Energy Waste Policy Act of 2005. AMD is not exempt from the Safe Drinking Water Act, while HFFF is for naturally occurring contaminants. The Pennsylvania Environmental Good Samaritan Act could potentially eliminate liability of parties responsible for the spread of AMD into uncontaminated water sources though hydraulic fracturing operations. It is unclear who would be responsible for compensating stakeholders over damaged water sources while utilizing AMD for hydraulic fracturing operations.

5.3 Potential Challenges for Implementation

Challenges when utilizing AMD for hydraulic fracturing operations mainly come in the forms of infrastructure, regulations, enforcement, cost, and innovation. If hydraulic fracturing operations decide to utilize AMD it will require construction of more storage and treatment facilities. Newly constructed facilities will need to follow regulations requiring investment and inspections. Future innovations could potentially deem new facilities useless.

Treated AMD yielded better HFFF treatment results in the Kondash et al. (2013) and ProChemTech (2009) studies, which means that hydraulic fracturing operations would focus on using treated AMD. Untreated AMD would not meet the water quality impoundment parameters recommended by the PADEP (2011). Many forms of treated AMD would not meet impoundment requirements, but treated AMD would be the only possible form to meet storage requirements. Taking treated AMD for hydraulic fracturing from existing treatment facilities will not bring relief to watershed stakeholders. PADEP (2011) states that liability exclusion will only be granted if parties contribute to remediation costs. This restriction would mean the construction of new AMD treatment facilities which would require time, money, and oversight.

Projects such as these could increase the cost of drilling for hydraulic fracturing. The AMD would then have to be transferred to storage sites requiring construction and inspection.

It is also unclear whether the Bevill Amendment or the Energy Policy Act of 2005 would apply to storage facility classifications. If the exemptions apply, then AMD used for hydraulic fracturing operations would meet requirements for all storage facilities. Without the exemptions, AMD for hydraulic fracturing would most likely only meet requirements for the centralized wastewater impoundment for mining and drilling. This scenario is the most expensive option, requiring inspections and locations not adjacent to drilling sites. New technologies have potentially made the practice of waterless hydraulic fracturing possible which could make the utilization of AMD less appealing to stakeholders (Goodman 2012).

5.4 Conclusions

Hydraulic fracturing continues to be a controversial industrial practice among water resource stakeholders and the utilization of AMD does not ease the polarizing conversation. AMD continues to impact thousands of miles of Pennsylvania streams with slow and expensive cleanup results. Allowing for AMD to be utilized for hydraulic fracturing creates a potential for drilling companies to partner with government agencies and nonprofits to invest in AMD treatment facilities, for all stakeholders to benefit from remediation. Investments and practices of utilizing treated AMD in hydraulic fracturing will save billions of gallons of uncontaminated water within the state of Pennsylvania, which supports multibillion dollar industries like agriculture and recreational fishing. Mixing treated AMD with HFFF will remove contaminants from both forms of industrial wastewater which could save time and money for wastewater treatment plants while lowering the volumes of contaminated discharges into surface waters.

The presence of AMD in hydraulic fracturing operations could accompany relief for some water sources, with potential increased degradation of others. Aquifers, surface waters, soil, and vegetation in close proximity to AMD utilization and storage sites are at an increased risk than before. Spills of AMD, followed by adsorption, erosion, and runoff could contaminate watersheds adjacent to hydraulic fracturing sites. Utilizing AMD for hydraulic fracturing could potentially contribute to the spread of AMD impacts around the state.

Liability and compensation recommendations for potential damages from utilizing AMD are ambiguous. The Environmental Good Samaritan Act could eliminate civil liability of hydraulic fracturing parties utilizing the AMD, leaving stakeholders with damaged health and property without compensation. Applications of the industrial waste exemptions from the Energy Policy Act of 2005 and the Bevill Amendment could potentially determine liability and wastewater storage facility qualifications recommended by the PADEP (2011).

The practice of utilizing AMD for hydraulic fracturing could increase the cost of conducting business for drilling companies. Investing in treatment plants, constructing storage sites, inspection fees, and transporting AMD will not be cheap. Potential expenses for utilizing AMD could deter drilling companies from implementing the industrial practice. Waterless hydraulic fracturing has been researched and practiced with successful results. If waterless hydraulic fracturing were to be implemented on a mass scale, then time and money invested in AMD utilization storage, transportation, treatment, legislation, and liability could become worthless (Goodman 2012).

5.5 Recommendations

Many concerns and questions regarding the utilization of AMD in the hydraulic fracturing process are unknown due to a lack of studies and monitoring of this practice in the field. The studies conducted on utilizing AMD in hydraulic fracturing are limited and have primarily been conducted in the lab. PADEP should make procedural recommendations for HFFF and AMD mixing. Mixing should happen outside of the wellbore to prevent scaling from mineralization.

To further understanding of the benefits and risks associated with the industrial practice, more studies should be conducted in the field. Hydraulic fracturing sites that have been utilizing AMD should be monitored for soil, surface water, and aquifer contamination to provide a better understanding of environmental risks and best practices. Analyzing recycled HFFF and AMD mixtures could provide data about treatment benefits and complications, since the lab studies did not analyze repeatedly recycled mixtures. This type of study could also provide information about possible wellbore damage.

Potentially increased costs of utilizing AMD for hydraulic fracturing businesses need to be calculated. Permits, treatment, transportation, and construction of facilities could shrink profit margins for hydraulic fracturing businesses, which could negate the utilization benefits. Liability and waste exemptions need to be clarified by state regulators and legislatures. This determination of liability will help calculate potential costs for all stakeholders so they can properly prepare for legal challenges and risks before utilizing AMD on a mass scale. Research regarding the possible mass scale practice of waterless hydraulic fracturing needs to be thoroughly investigated to prevent wasted time and resources.

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