

An Analysis of the Impact of Mountain Top Removal Mines on  
Private Drinking Water Wells in West Virginia

By

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***Abstract***

Mountaintop removal coal mining (MTR) is currently the dominant driver of landuse change in the central Appalachians. It involves the clearing of forests, removal of topsoil, and use of explosives to remove the overburden above the coal seams. After mining is complete, some of the overburden is replaced and the excess is pushed into adjacent valleys. These valleyfills bury headwater streams and generate mine drainage, which contains elevated concentrations of sulfate and trace metals and metalloids with known toxicity. Numerous studies have reported that residents of counties where MTR occurs experience disproportionate levels of adverse health effects including increased rates of cancer mortality and birth defects. In this study, the link between MTR and community health was investigated by sampling and analyzing private drinking water wells and using geospatial statistical models to determine whether MTR is affecting drinking water quality. Over 30% of the wells sampled had concentrations of aluminum, manganese, and iron high enough to cause bad taste and staining. However, these concentrations were not correlated with distance from mining activity. Wells downstream of mines were found to have higher levels of selenium, uranium, sulfate, nitrate, and potassium, but none of these contaminants were present in concentrations that exceeded drinking water standards.

## ***Introduction***

### *Importance of Coal*

Approximately half of the electricity consumed in the United States since 1995 has been produced from coal and the demand for coal-derived electricity is projected to grow 1.3% per year from 2005 to 2030 (Epstein et al. 2011). To meet this demand, over a billion tons of coal was produced in the United States in 2010. Some of the most efficient coal in the country is found in the mountains of the Appalachian coalfield region spanning West Virginia, Virginia, Kentucky, and Tennessee. This coal produces more energy when burned than western coal, while producing less air pollution due to its low levels of sulfur. West Virginia is the second largest producer of coal behind Wyoming, having produced over 144 million tons in 2010 (Coal Facts 2010). Approximately 40% of this coal was produced through mountaintop removal mining (Coal Facts 2010). Mountaintop removal (MTR) has now been performed on at least 500 peaks, burying, according to EPA predictions, almost 4,000 Km of headwater streams by 2012 (The Effects of Mountaintop Mines and Valley Fills on Aquatic Ecosystems of the Central Appalachian Coalfields).

### *Description of MTR*

MTR emerged in the late 1960s and gained popularity when the Clean Air Act amendments passed in 1990 drove up demand for low sulfur coal (Holzman 2011). It is now the dominant driver of land-use change in the central Appalachian eco-region (Saylor 2008). There are several phases to MTR: first the forests above the coal seam are cleared and the topsoil is

removed; next dynamite is used to dislodge the rock above the coal seam; the coal is then removed using large machinery. Once the coal has been removed and mining is complete, the overburden is used for re-grading or pushed into adjacent valleys creating valley fills and permanently burying headwater streams (Lindberg et al. 2011).

### *Adverse Effects on Surface Water*

There is a growing body of evidence of the adverse effects of valley fills on streams. In some cases these effects are still observed nearly two decades after the mines have been reclaimed (Palmer et al. 2010; Pond et al. 2008; Lindberg et al. 2011). Valley fills contain rock that is much more loosely packed and with much smaller particle sizes than the bedrock that formed the mountain before it was mined. This loosely packed rock has a much larger surface area to be exposed to air and water. When the coal and overburden are exposed to oxygen and moisture, iron-disulfide minerals such as pyrite ( $\text{FeS}_2$ ) become oxidized, which can result in increased levels of sulfuric acid, iron, manganese, and aluminum in both surface and groundwater (McAuley and Kozar 2006). The increased acidity can furthermore dissolve calcareous rock causing increased levels of calcium and magnesium. Pyrite oxidation is the principle driver of acid mine drainage, which is a significant concern in northern parts of West Virginia. In southern West Virginia, however, the calcareous strata and lower concentrations of sulfur in the coal tend to result in alkaline mine drainage (Pond et al. 2008). Adverse effects of alkaline mine drainage on surface waters can include increased conductivity, sulfate, and selenium, resulting in impaired biological conditions (Pond et al. 2008). Increased selenium is particularly important for ecological health, because it causes malformations in developing fish

and a loss of sensitive aquatic biota (Bernhardt and Palmer 2011; Mountaintop Mining/Valley Fills in Appalachia: Final Programmatic Environmental Impact Statement; Pond et al. 2008).

### *Adverse Effects on Surrounding Communities*

A variety of studies have also been published investigating the relationship between MTR and adverse health effects in the surrounding communities. The coal fields of southern West Virginia are characterized by high rates of poverty, high rates of renter occupancy, and low levels of education (West Virginia Counties 2009; Halverson 2004). In a study using human health data at the county level, the rates of “other” cancer and total cancer rates were found to increase as distance from mines decreased (Hendryx, Fedorko, and Anesetti-Rothermel 2010). Higher rates of birth defects and respiratory cancer have also been found in counties with higher levels of mining activity in addition to lower self-rated health (Ahern et al. 2011; Hendryx 2009; Zullig and Hendryx 2011).

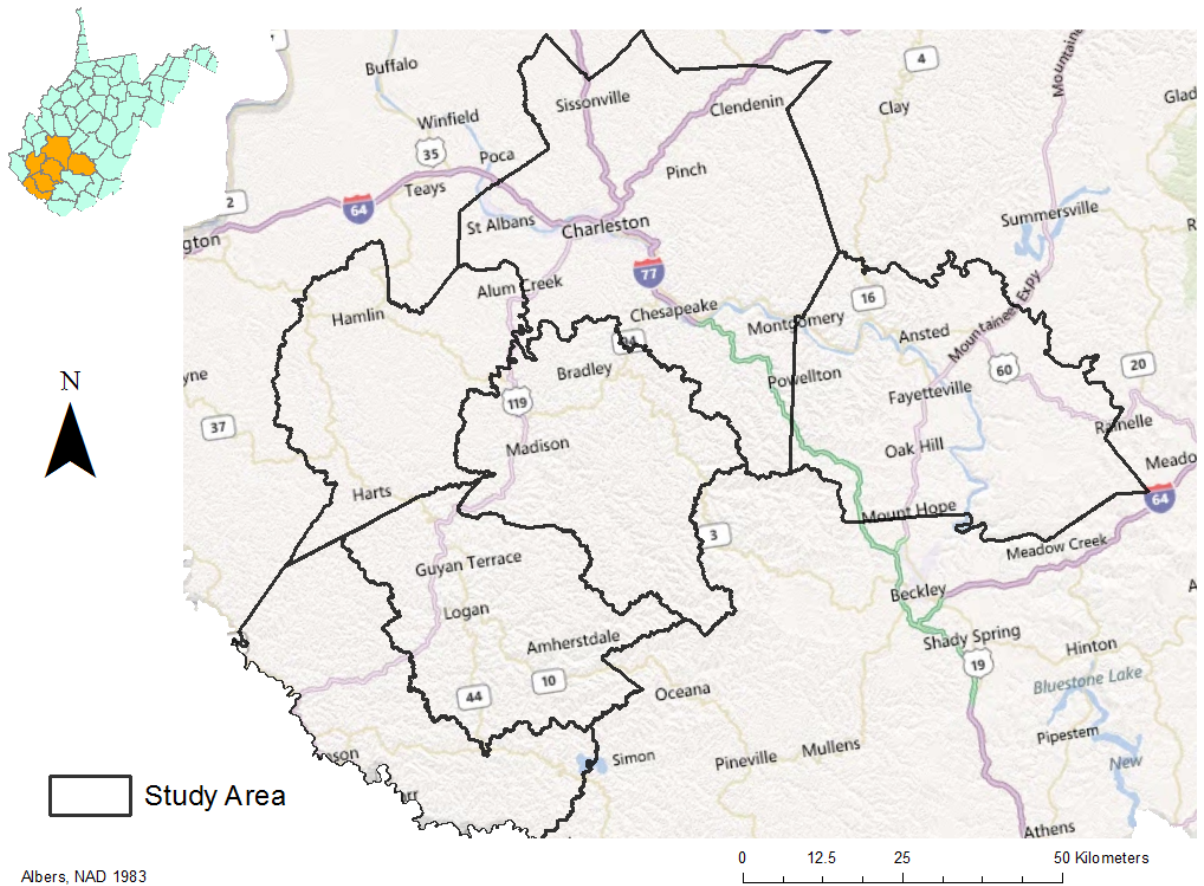
There are several routes of exposure by which MTR practices could be affecting the health of the surrounding communities. One route is through the production of particulate air pollution, which is correlated with increases in lung cancer as well as cardiopulmonary mortality (Pope et al. 2002). Another possible route is through the drinking water. Many inhabitants of rural West Virginia have no access to city water and therefore rely on private wells as their primary water source. Unlike public drinking water sources, private wells are not regulated by the state or federal government. Contamination of these wells as a result of mining activity could potentially be causing some of the adverse health outcomes observed in previous studies

(Hendryx 2009; Hendryx and Ahern 2008; Hendryx, Fedorko, and Anesetti-Rothermel 2010; Ahern et al. 2011; Zullig and Hendryx 2011).

## **Methods**

### *Study Area: Demographics*

Six counties in southern West Virginia were selected as the focus for this study because of the high rates of MTR: Boone, Fayette, Kanawha, Lincoln, Logan and Mingo. This area is characterized by high rates of poverty and renter occupancy (West Virginia Counties 2009). These characteristics are important because it makes it more difficult for residents to purchase filters or obtain other sources of water once their wells have been contaminated.

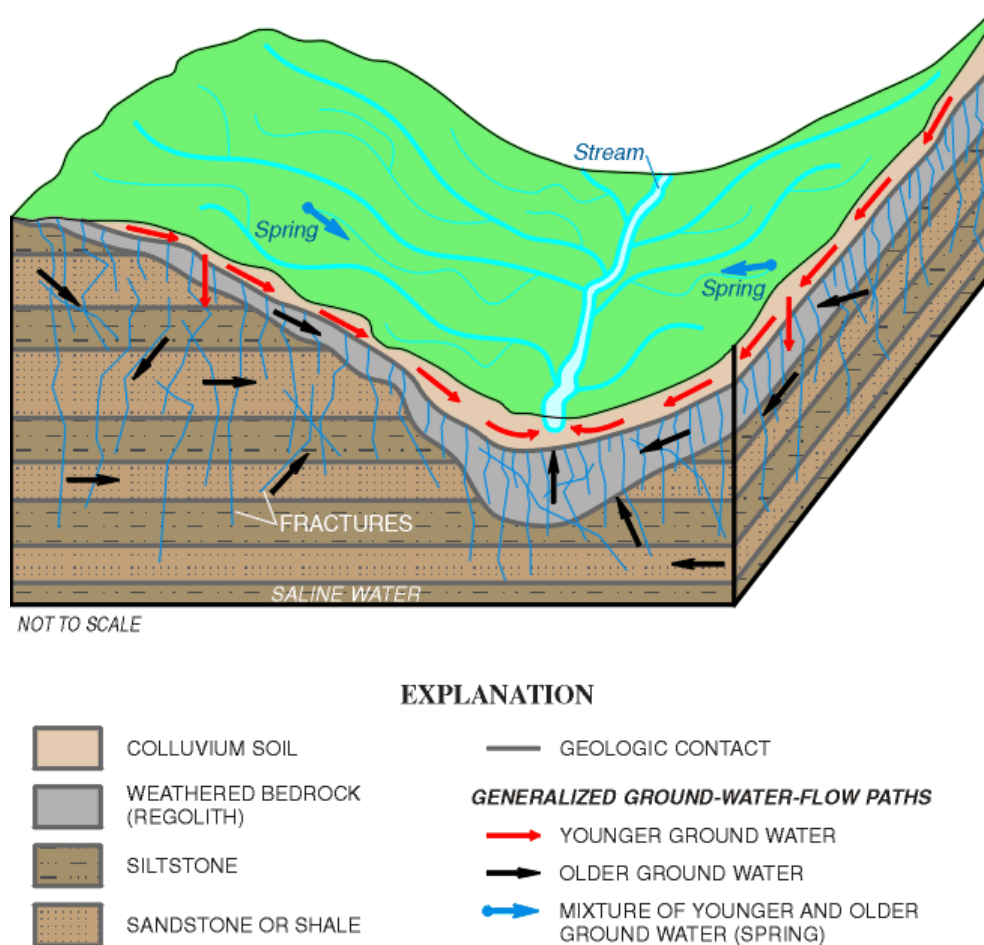


**Figure 1: The study area**

*Study Area: Hydrogeology*

Our study area is part of the Lower Pennsylvanian Sedimentary Bedrock Aquifer, which consists of fractured sandstone, limestone, coal, shale, and siltstone. This type of aquifer is classified as a clastic aquifer in contrast to a carbonate aquifer, which consists of limestone and dolomite. Kanawha County also contains the Kanawha River Quaternary age alluvial aquifer, but none of our samples are within this aquifer. Most of these rock types are permeable to a depth of about 100ft, but only coal seams are consistently permeable at depths greater than 200ft

(Messinger et al. 2000). Because of the low permeability of the rock and the steepness of the terrain, ground water in the Appalachian Plateaus and Blue Ridge Physiographic Provinces moves mostly in a network of narrow fractures within a few hundred feet of the land surface.

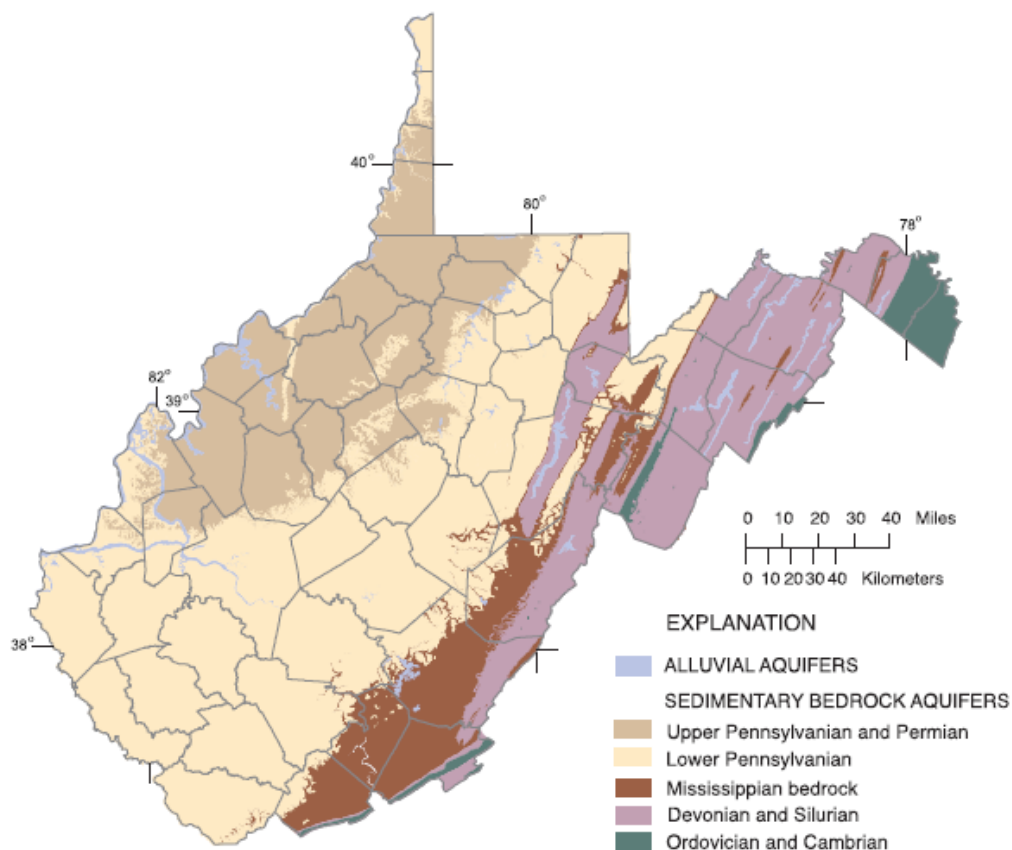


**Figure 2. Conceptual ground-water flow in a fractured-rock setting (Harlow et al. 1991)**

As in the majority of landscapes, surface and ground water systems in this area are inter-connected. Seepage from precipitation and surface water recharges groundwater when groundwater levels are lower than surface water. Groundwater, in turn, rejoins surface water in streams and springs. The movement of both surface and ground water is primarily controlled by the land-surface form and geology of an area (Winter et al. 1998). In confined aquifers



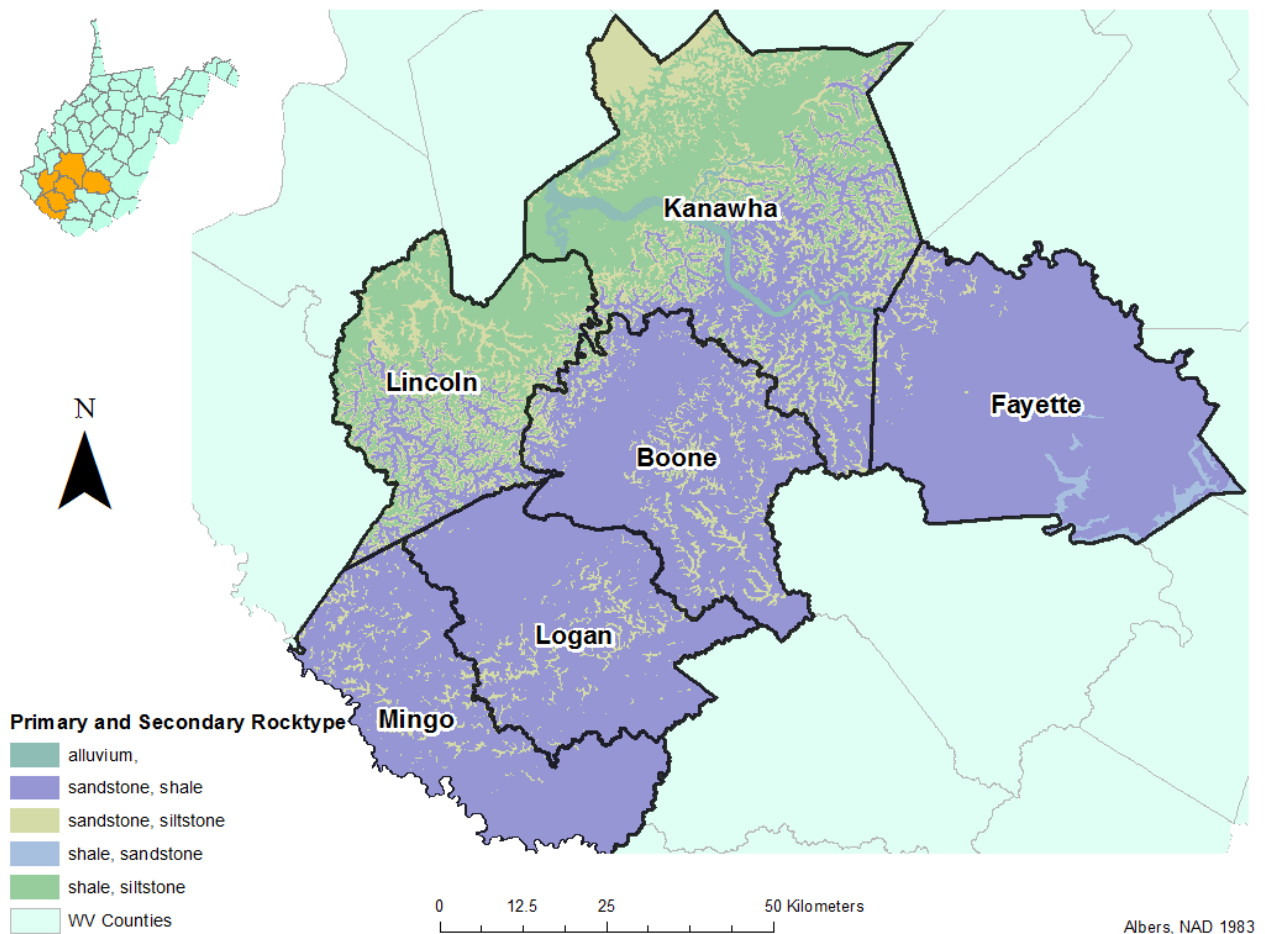
groundwater can flow against the slope of local topography. These aquifers, when tapped, create artesian wells. In our study area, however, groundwater flows through unconfined aquifers and like surface water drains toward the nearest stream. In general, each local aquifer is bounded by the surrounding ridge tops which isolate it and protect it from all but local contaminant sources (Paybins, National Water-Quality Assessment, and Geological 2000).



**Figure 3. Major hydrogeologic units in West Virginia (Kozar et al. 2001)**

Groundwater flow, while generally in the same direction as surface water flow, is largely controlled by three factors – recharge, surface topography, and hydraulic conductivity of the material through which the groundwater flows. Mining, both surface and underground can

drastically alter each of these factors. Groundwater recharge can be changed depending on how the surface material is handled and how the site is re-vegetated. Surface topography can be altered depending on the final reclamation contours, and hydraulic conductivity of the overburden is permanently altered by the mining process, in fact the mine spoil may be several orders of magnitude more conductive than the parent material (Brady 1998).



**Figure 4: Geology of the study area**

*Study-Area: Other Potential Contamination Sources*

This area is affected not only by MTR but also by extensive underground mining and other coal processing activities. After the coal is mined, it is processed in preparatory plants. One of the byproducts of this processing is coal slurry waste. Coal slurry waste is disposed of by either storing it in impoundments or injecting it into old underground mines. In assessments of these waste sites, the US EPA and USGS have identified high levels of arsenic, cadmium, chromium, iron, lead, zinc, manganese, and sulfate in the coal slurry (Wigginton et al. 2008). Underground mining has also been associated with changes in both surface and ground water chemistry most notably through acid mine drainage, but also through alkaline mine drainage containing elevated levels of sulfate and other ions (Sams 2000).

*Data Sources*

Mining activity data was obtained from the West Virginia Department of Environmental Protection (WVDEP) and Skytruth Inc. Permits for all mines, slurry impoundments, and coal processing plants are available from the WVDEP. Valley fill sizes, locations, and ages were also obtained from the WVDEP. Locations of slurry injection sites were obtained from Evan Fedorko at West Virginia University. Permitted areas are not always fully mined so the surface mining activity delineated by SkyTruth, Inc was also used. Surface mining activity was determined by SkyTruth Inc. with a decision tree analysis using Landsat satellite imagery to isolate areas of mining activity to serve as an account of historical occurrences of mountaintop mining from 1976 to 2005.

In order to develop a sampling strategy, point locations of all the buildings in our study area were obtained from the West Virginia GIS Technical Center. This data was collected from

2003 natural color orthophotography by the WV Statewide Addressing and Mapping Board. To determine the location of households with private drinking water wells, well completion permits from 1990-2010 were obtained from the WV Bureau for Public Health and entered into a Microsoft Access database. Other geospatial data layers utilized throughout this analysis included elevation, flow accumulation, and flow direction grids at a 30m resolution obtained from the National Hydrography Dataset Plus (NHD Plus).

#### *Recruitment of Volunteers and Collection of Well Water Samples*

Several factors prevented us from obtaining a random sample of residents that used private drinking water wells. Tax parcel data was not available for the majority of counties in our study area. Instead well completion permits were used to locate and contact residents using private drinking water wells. However, physical address data was missing for over half of the records, causing the actual location of the wells to be unknown.

Instead of conducting a random sample of well users, residents were grouped into communities and target communities were selected based on proximity to different types of mining activity. A data layer of all of the buildings in our study area was used to determine the approximate location of each community. A one-mile buffer was created around surface mine permits, underground permits, and coal slurry impoundments or injections. Any communities with the one-mile buffer of a surface mine was considered potentially affected by surface mining. Any community within a one-mile buffer of an underground mine was considered potentially affected by underground mining and any community within a one-mile buffer of a coal slurry injection site or impoundment was considered potentially affected by coal slurry. A

one-mile buffer was used because that is the radius used for the source water protection area around wells that are used for public drinking water sources.

To recruit volunteers we mailed letters to individuals living in selected target communities using information from the well completion reports. We also relied heavily on community partners, including the county health departments, to distribute information about the study to residents who they thought might be affected and were interested in having their wells sampled.

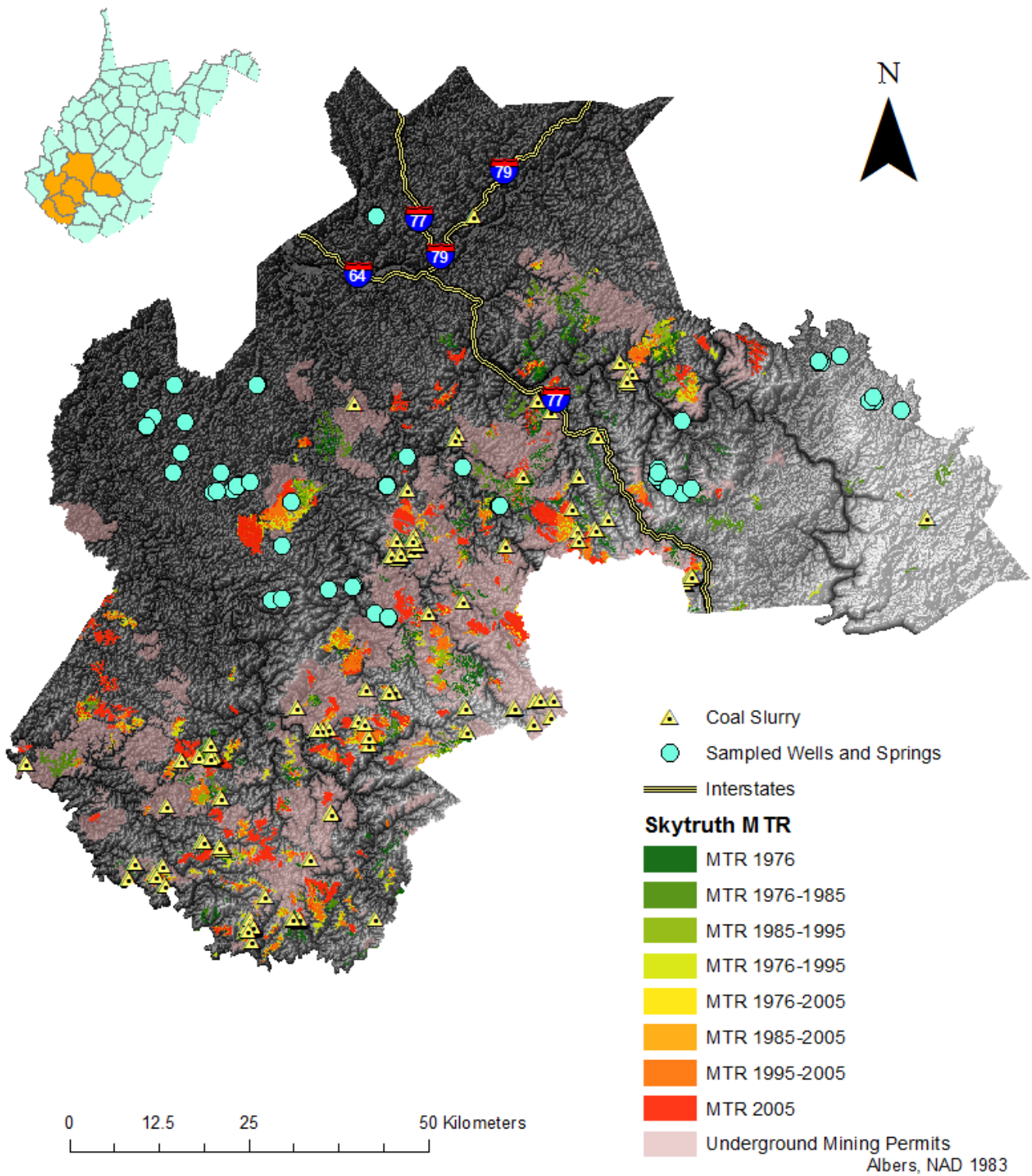
All participants were surveyed before their well or spring was sampled. Information was obtained on age and depth of the well, whether it was drilled or dug, whether animals were kept on the property, how much well water was drunk on an average day, and what type of treatment they used if any. Wells were purged to remove stagnant water, and then monitored for pH, conductivity, dissolved oxygen, and temperature until stable values were recorded. Samples were collected as close to the well or spring as possible and from the kitchen tap. A suite of major and trace elements were measured including  $\text{SO}_4^{2-}$ , magnesium, calcium, iron, manganese, arsenic, lead and selenium. Samples for metals and major ions were filtered with a 0.45- $\mu\text{m}$  syringe disk filter. Samples from the kitchen tap were not filtered. Dissolved metals were preserved with 0.2% v/v trace metal grade nitric acid with all sample bottles placed on ice prior to storage at 4.0 °C at the laboratory. Samples were also collected to measure carbonate levels and isotope ratios at a later date.

**Table 1: Parameters measured**

Type	Parameters
Well Attributes	Depth, Age, Distance to nearest surface water, Animals kept on the property
Field Measurements	Conductivity, Dissolved Oxygen, pH, Temperature,
Anions	Sulfate (SO <sub>4</sub> <sup>2-</sup> ), Nitrate (NO <sub>3</sub> <sup>-</sup> ), Bromide (Br <sup>-</sup> ), Chloride (Cl <sup>-</sup> ), Fluoride (F <sup>-</sup> )
Cations known to adversely affect human health	Arsenic (Ar, Lead, Chromium, Cadmium, Thallium, Uranium, Barium, Beryllium, Selenium
Cations known to affect taste, odor, or color of water, can stain skin or teeth	Aluminum, Copper, Iron, Manganese, Silver, Zinc
Cations useful for isotope labeling	Boron, Strontium
Other cations	Calcium, Cobalt, Potassium, Lithium, Magnesium, Molybdenum, Nickel, Sodium, Thorium, Vanadium
Samples for future measurements	Carbonate, Isotope Ratios

## Results and Analysis

Groundwater samples from fifty-eight wells and springs in Lincoln, Boone, Kanawha, and Fayette counties were collected and analyzed between October 2011 and March 2012. Eight of the sampled wells and springs had treatment systems upstream of where the sample was taken and were not included in the analysis. Cation results were obtained for 40 groundwater samples that were upstream of all treatment systems. Anion results were obtained for 48 groundwater samples that were upstream of all treatment systems.



**Figure 5: Locations of mining activities and sampled wells in the study area.**

Manganese, iron, and aluminum all exceeded the EPA secondary maximum contaminant level standards in over 30% of the filtered samples collected (23, 20, and 16 samples respectively). While manganese, iron, and aluminum, are not known to cause adverse health effects, they can cause bad odor and taste as well as staining. None of the filtered samples exceeded the maximum contaminant limits for arsenic, lead, cadmium, uranium, or selenium, although four of the unfiltered samples collected from the kitchen faucet were above the MCL for lead and one was above the MCL for arsenic. None of these wells that exceeded these limits were downstream of mines. Selenium levels above 5ppb, the EPA limit for stream water quality, were only found in 3 springs in close proximity in Fayette County. Correlations were observed between both iron and manganese and between calcium and magnesium indicating that they were coming from similar sources. Conductivity was most strongly correlated with sodium but was also slightly correlated with calcium and was not correlated with sulfate.

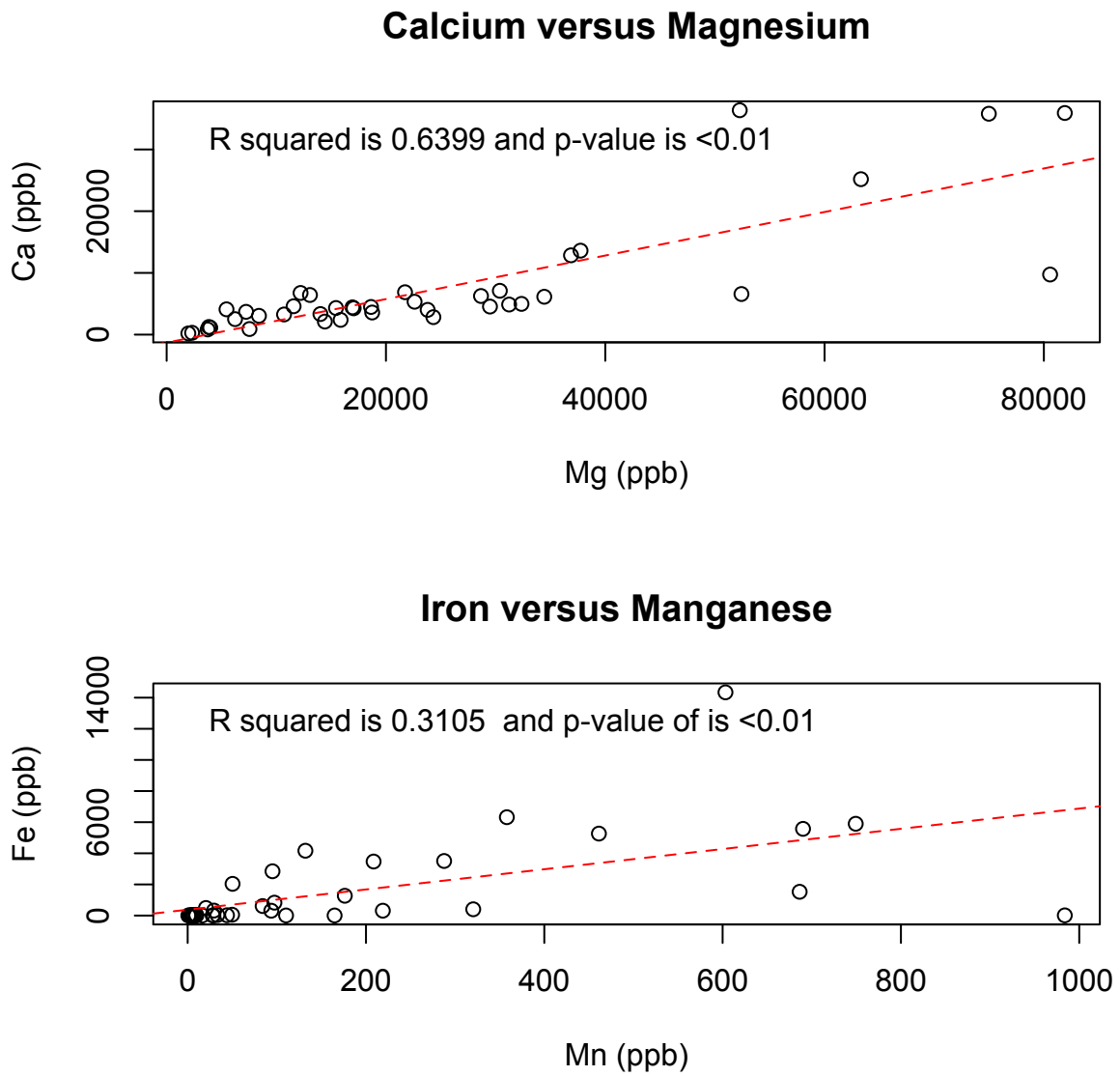
**Table 2: Summary statistics of results.**

Field Parameters	Mean	Standard Deviation	Minimum	Median	Maximum	# samples
Dissoved Oxygen (ppm)	2.74	3.12	0.10	1.26	10.67	56
pH	6.92	0.82	4.77	6.91	8.95	56
Temperature C	13.41	1.44	10.00	13.45	16.40	58
Conductivity (uS)	325.92	400	0.76	247.50	2800.00	57

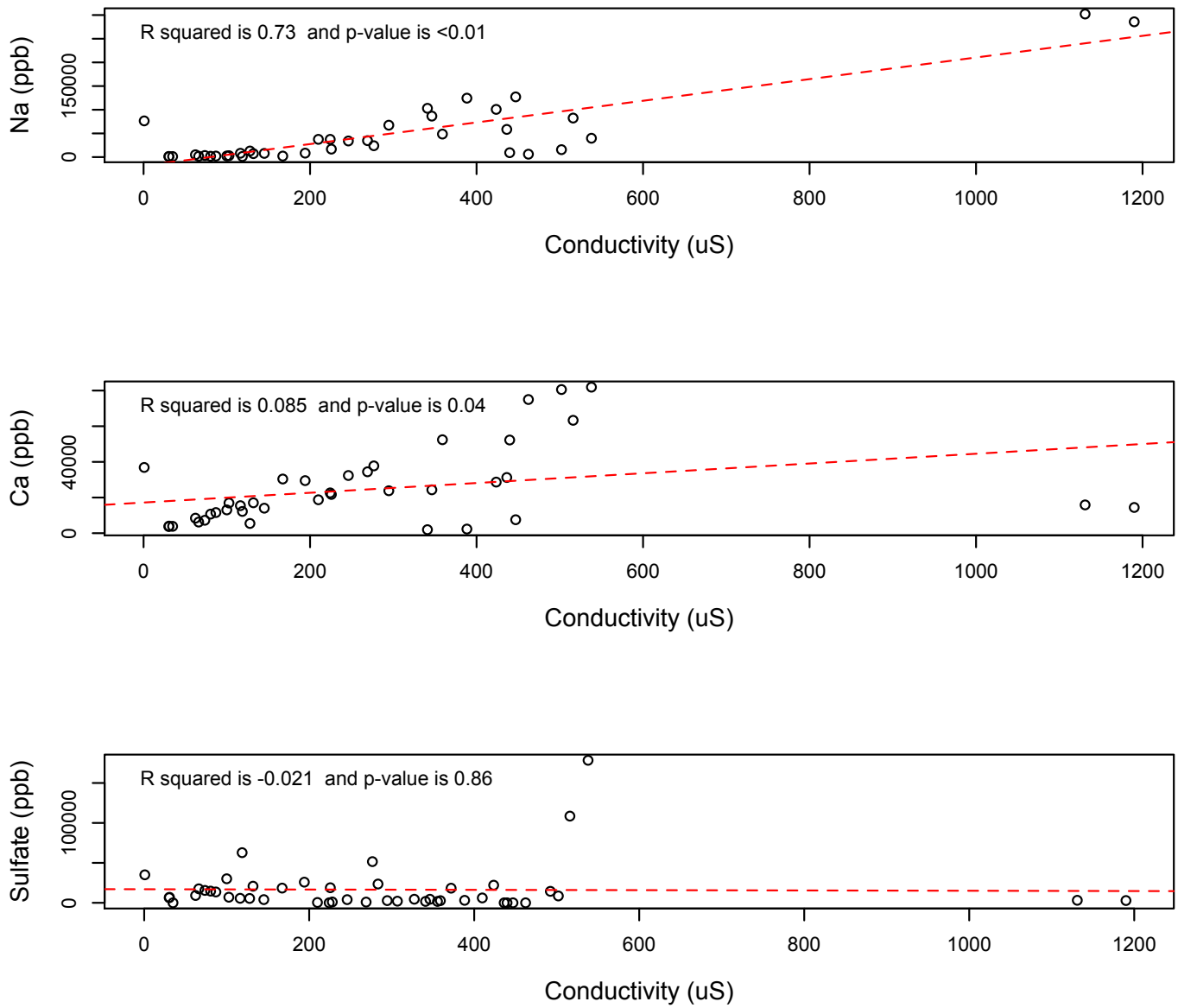
Anions (ppb) (n=56)	Mean	Standard Deviation	Minimum	Median	Maximum
Br	0.10	0.42	BDL	BDL	2.64
NO3	0.46	1.03	BDL	BDL	5.15
Fl	0.22	0.34	BDL	0.08	2.06
SO4	16.13	28.82	BDL	6.24	178.40
Cl	37.03	106.06	BDL	6.50	686.09



<b>Cations (ppb) (n=51)</b>	<b>Mean</b>	<b>Standard Deviation</b>	<b>Minimum</b>	<b>Median</b>	<b>Maximum</b>
<b>Ag</b>	BDL	0.02	BDL	BDL	0.05
<b>Al</b>	56.68	52.15	BDL	42.40	348.55
<b>As</b>	0.47	1.37	BDL	0.10	9.45
<b>B</b>	34.66	41.61	BDL	17.30	179.95
<b>Ba</b>	170.78	210.08	BDL	62.20	867.90
<b>Be</b>	0.06	0.21	BDL	BDL	1.20
<b>Ca</b>	19464.56	21182.07	BDL	14023.30	81909.00
<b>Cd</b>	0.05	0.23	BDL	BDL	1.65
<b>Co</b>	0.65	3.41	BDL	BDL	24.15
<b>Cr</b>	0.22	0.22	BDL	0.15	0.95
<b>Cu</b>	10.44	31.74	BDL	0.60	175.45
<b>Fe</b>	1136.84	2559.26	BDL	34.10	14341.10
<b>K</b>	2229.10	1750.55	BDL	2011.20	9463.05
<b>Li</b>	11.13	17.80	BDL	5.10	84.20
<b>Mg</b>	5980.42	8734.96	BDL	4011.05	36355.65
<b>Mn</b>	134.30	229.79	BDL	28.90	983.80
<b>Mo</b>	0.10	0.22	BDL	0.05	1.40
<b>Na</b>	45134.01	62181.24	BDL	24018.75	302188.30
<b>Ni</b>	4.10	13.20	BDL	0.20	82.85
<b>Pb</b>	0.31	0.61	BDL	0.10	3.45
<b>Se</b>	0.50	1.69	BDL	0.05	9.30
<b>Sr</b>	308.15	448.16	BDL	90.05	2089.70
<b>Th</b>	BDL	0.01	BDL	BDL	0.05
<b>Tl</b>	BDL	-	BDL	BDL	BDL
<b>U</b>	0.01	0.05	BDL	BDL	0.35
<b>V</b>	0.23	0.14	BDL	0.20	0.75
<b>Zn</b>	50.30	96.30	BDL	21.60	554.95



**Figure 6: Comparing concentrations of various cations**



**Figure 7: Conductivity compared to individual ions**

*Modeling the Effects of Mines*

Once the samples had been collected, a more precise geospatial model was constructed in order to determine which wells were potentially affected by mining activities. The watershed of each well was delineated using a 30m digital elevation dataset (USGS National Elevation Dataset) and spatial analyst hydrology tools in ArcMap 10. In order to get a watershed of an appropriate size, the location of each well was snapped to the closest stream in the USGS National Hydrography Dataset and then to the closest pour point in 50m in the associated flow accumulation grid.

The resulting watersheds were then intersected with point locations of WVDEP mining permits. Only one of the wells sampled was downstream of a coal slurry permit. Fifteen wells and springs were downstream of both underground and surface mining permits, however two of these samples had been treated and weren't used in the analysis. The other 40 wells and springs were not downstream of any mining permits and were used as controls. Because all of the samples collected downstream of surface mines were also downstream of underground mines, we could not distinguish from this dataset which type of mining was more strongly correlated with changes in the chemistry of private drinking water wells.

A linear regression was performed to compare the concentrations of each measured parameter in the control wells to the concentrations in the wells downstream of mines. A log transformation was used because the measured parameters tended to be left skewed. A regression was run for each measured parameters, using the measured parameter as the dependent variable and a dichotomous variable indicating the presence of mining as the only predictor variable.

**Table 3: Comparison of wells downstream of mines and control wells using a log-linear regression with a dichotomous mining indicator variable.**

<b>Parameter</b>	<b>Ratio of average measurements at mined sites to average measurements at control sites.</b>	<b>p-value</b>
Selenium	15.85	0.00
Nitrate	9.90	0.05
Nickel	8.17	0.10
Cadmium	5.77	0.02
Sulfate	3.48	0.03
Uranium	3.45	0.01
Copper	2.93	0.33
Dissolved Oxygen	1.37	0.53
Beryllium	2.05	0.41
Sodium	1.79	0.34
Magnesium	1.65	0.22
Zinc	1.50	0.26
Potassium	1.42	0.05
Bromide	1.34	0.70
Boron	1.33	0.49
Lead	1.28	0.53
Aluminum	1.26	0.23
Strontium	1.25	0.67
Chloride	1.08	0.90
Calcium	1.05	0.88
Vanadium	1.05	0.79
Thallium	BDL	
pH	0.99	0.82
Thorium	0.77	0.42
Chromium	0.74	0.45
Conductivity	0.65	0.33
Silver	0.52	0.18
Cobalt	0.48	0.51
Barium	0.37	0.01
Lithium	0.32	0.40
Fluoride	0.28	0.02
Molybdenum	0.24	0.08
Arsenic	0.15	0.02
Manganese	0.07	0.00
Iron	0.04	0.00

Selenium, uranium, cadmium, sulfate, potassium, and nitrate were all elevated in the groundwater samples downstream of mines, significant at the 95% confidence level.

Manganese, iron, barium, arsenic, fluorine, molybdenum, and silver, on the other hand were all elevated in the control samples at the 95% confidence levels. The other parameters did not vary significantly between control samples and samples downstream of mining sites.

The size of the calculated watersheds of the wells varied tremendously (0.01 to 292 Km<sup>2</sup>). To adjust for this discrepancy and better understand the spatial relationship between well chemistry and proximity to mines, a Bayesian statistical model was constructed.

Each mine permit was considered a source that could affect well chemistry. I considered a spatial array of  $j = 1, \dots, J$  sources that contributed to  $i = 1, \dots, n$  wells depending on the effective distance  $d_{ij}$ . The effective distance was calculated as the Euclidean distance between the location of the mine permit and the location of the well. Only mine permits within the watershed of each well were considered. The concentration at well  $i$  was taken as the sum of these contributions plus observation error  $v_i$ ,

$$y_i = \sum_{j=1}^J f(x_j; \beta, \sigma^2) k(d_{ij}; \alpha) + v_i$$

$$v_i \sim N(0, \tau^2)$$

However, mine permits differ by type: underground versus surface, and by age, so the source strength was allowed to vary. The source strength depends on predictors  $x_j$  with fitted parameters  $b$  and error  $\varepsilon_j$ ,

$$f(x_j; \beta, \sigma^2) = \mu_j + \varepsilon_j$$

$$\mu_j = x_j \beta$$

$$\varepsilon_j \sim N(0, \sigma^2)$$

The kernel describes the decay in influence of each mine on each well with effect distance  $d_{ij}$

with fitted parameter  $\alpha$ :

$$k(d_{ij}; \alpha) = \exp\left(-\left(\frac{d_{ij}}{\alpha}\right)^2\right)$$

Then the length- $n$  vector of response concentrations is:

$$N_n(y|K\mu, \Sigma)$$

$$\Sigma = KK^T \sigma^2 + I\tau^2$$

The variance  $\tau^2$  imposed at location  $i$  is termed a 'nugget' in the geospatial literature. It allows for error in observations, and it is needed to make the covariance matrix invertible. We would like to estimate the kernel parameter ( $\alpha$ ) and the fitted parameters for the predictors of source strength ( $\beta$ ). To obtain these values we will need to estimate  $\sigma^2$  and  $\tau^2$ . A Gibbs Sampler was used to estimate  $\beta$ , used to predict the strength of the effect from each individual mine permit,  $\alpha$ , the effect of distance, and  $\tau^2$ , the variance at location  $i$ .

The model was fitted to each of the parameters that were positively correlated with mining activity. The model did not converge to a posterior distribution for  $\alpha$  for any of the measured parameters, indicating that there is no spatial relationship between the mine

locations and the concentration of contaminants in the wells. This result contradicts the results of our earlier analysis, which found statistically significant differences between the wells that were downstream of mining activity and those that weren't. The discrepancy between the results of these two analyses indicate that although there are differences between the wells downstream of mines compared to controls, the Euclidean distance from mining permits cannot be used to predict the concentration of any of the measured parameters in the groundwater. This could be due to the fact that the mines actually aren't affecting the groundwater and the differences found in the first analysis were due to other factors such as geology. Alternatively, the failure of the model to detect a spatial correlation could be due to invalid assumptions made when constructing the model. The model was created with the assumption that the measured sample concentrations were normally distributed, although they were actually substantially left skewed. Additionally, using point locations of mine permits may not be an accurate method of representing mines that can cover many hectares. Finally, our sample size may have been too low compared to the number of mines to determine a correlation.

### ***Discussion***

Two previous USGS studies have sought to determine the effects of surface mines on private water wells. A USGS study conducted in 2000 (Paybins, National Water-Quality Assessment, and Geological 2000) compared 58 wells sampled near coal surface mines and 28 wells in unmined areas using a simple linear distance from reclaimed surface mines to determine the relationship between surface mining and a variety of ions. They found that



sulfate concentrations in ground water were higher than background levels in shallow wells within 1,000 feet of reclaimed surface mines and that iron, manganese, and aluminum were higher than background concentrations within 2,000 feet of reclaimed surface mines. The second USGS study conducted in 2006 (McAuley and Kozar 2006) used a more precise sampling strategy. They made the assumption that the principal direction of groundwater flow coincided with topographic gradient and sampled 58 wells within approximately 2,000ft and downgradient from reclaimed surface mines whose bond had been released 2-12 years previously. They considered 2-years to be the minimum amount of time required for ground water to return to steady-state equilibrium after the reclamation process has been completed. In contrast to our study, they included wells in the high-sulfur coal regions of northern West Virginia as well as the low sulfur coal regions in our study area. They compared measured concentrations of mine-drainage-related constituents with distance to mines and with measured concentrations in 25 wells in unmined areas. They found significantly higher ( $p$ -value $<0.05$ ) concentrations of aluminum, ammonia, dissolved organic carbon, fluoride, hardness, iron, manganese, potassium, radon, silica, sulfate, and zinc in mined sites compared to unmined sites. However, at distances greater than 1,000 ft they found the concentration were typically at or below background concentration. They didn't find significant differences in barium, chloride, chromium, cobalt, copper, dissolved oxygen, nickel, nitrate plus nitrite, sodium, or total dissolved solids. Unlike the 2006 USGS study, in our comparison of wells at mined sites to control wells, we found that manganese was higher in unmined areas while zinc and aluminum did not vary significantly between mined areas and unmined areas. We did find elevated levels of sulfate, potassium, and nitrate in wells downstream of mines.

## Conclusion

Through this study, I was able to contribute to the current literature about the effect of MTR on groundwater. By calculating the watershed of each well and accounting for underground mining and coal slurry, and using Bayesian statistical modeling, I was able to add a level of complexity that wasn't present in previous studies. I did find statistically significant differences between wells downstream of mines and control wells. In some cases these differences were supported by the results of the 2006 USGS study (sulfate and potassium), but in other cases our results were directly contradictory (iron and manganese).

I was not able to establish a spatial correlation between concentrations in the groundwater and Euclidean distance to mining permits upstream using the Bayesian statistical model. As discussed earlier, this could either be due to the fact that mining activity is not affecting well water quality or to invalid assumptions made when constructing the model. The discrepancies in results both between different methods of analysis and between our results and those of previous studies demonstrate the need for further research on this topic. More samples as well as additional statistical models are needed to fully understand the effects of mountaintop mining on private drinking water wells. Additionally, more research is needed both on other potential exposure routes such as air quality and on health effects using household level data instead of county level data in order to determine the impact of MTR on the surrounding communities.

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