SECTION VII: Effects of Mine Subsidence on Streams during the 4th Act 54 Assessment

VII.A – Overview

This section assesses the impact of mine subsidence on the flow and biological health of streams in Pennsylvania. The University begins by comparing PADEP's multi-metric index for stream biology to other state and regional indices. There has been much discussion in recent years regarding the suitability and accuracy of PADEP's index, so here the University statistically evaluates its utility relative to similar indices. The University also recognizes that isolating the effect of mine subsidence on streams is a challenging task because many factors outside of mining influence stream ecosystems. Therefore, this section includes a discussion of the factors that cause natural variation in stream flow and biology, and highlights approaches to control for these factors in subsequent analyses. Following this discussion, the University reports the length of streams undermined and impacted during the 4th assessment period. PADEP's methodology for tracking stream impacts is discussed and the University highlights how the tracking procedures have changed since the 3rd assessment. The University then analyzes the effects of mining-induced flow loss and pooling on stream macroinvertebrate communities. The mitigation techniques that are utilized to restore impacted streams are described and the University reports the degree to which these were employed during the reporting period. The University also analyzes the effectiveness of mitigation in restoring communities to their pre-mining state. This section concludes with a comparison of pre- and post-mining Total Biological Scores for sites within the University's focal watersheds.

VII.B – PADEP Metric for Determining the Effects of Mining on Stream Biology

To determine if a stream is attaining its designated aquatic life status (Section I.E.4), the PADEP developed a macroinvertebrate community index known as the Total Biological Score (TBS; PADEP 2005a). This index serves as an indicator of stream biological integrity. The protocol for collecting macroinvertebrate samples and generating a stream reach's TBS is contained in Technical Guidance Document 563-2000-655 (hereafter TGD 563-2000-655; PADEP 2005a). TGD 563-2000-655 is based on recommended approaches in Barbour et al. (1999), the U.S. Environmental Protection Agency's (U.S. EPA) report on best methods for bioassessments. The two key elements of this approach are the development of a standardized sampling approach and the construction of an index that reliably reflects a stream's biological condition. To construct the index, scientists identify attributes (i.e. metrics) of the macroinvertebrate community that are ecologically relevant and reflect the community's response to pollution or other stressors. Typically, there is considerable redundancy among these metrics, i.e. many are highly correlated with each other. While multivariate statistics, such as principal components analysis, can remove redundancy and maximize the use of the information contained in all metrics, most states have settled on the use of indices that combine a small number of core metrics. The core metrics that jointly capture much of the necessary information are aggregated into a multi-metric index of stream health.

PADEP's development of the TBS index is recorded in an unpublished draft document (PADEP 2005b). Therein, the PADEP research that established the suitability of the TBS as a metric is documented. In addition, this study established a TBS benchmark above which the TBS is thought to indicate aquatic life use attainment and below which streams are deemed to be not attaining use. The benchmark for southwestern Pennsylvania is 50.1 (PADEP 2005b). While this document is unpublished, it is widely used by mine operators for comparing samples to the benchmarks that the document recommends for attainment. Nearly nine years after the draft version was produced, a final document has yet to be approved and disseminated to the public.

Since the development of the TBS, questions have been raised regarding its suitability for accurately assessing stream health and use attainment in southwestern Pennsylvania streams. These questions likely stem in part from the lack of a rigorous published report on the methodology used to develop and test the TBS. To partially assess the utility of TBS in measuring stream health, the University compared TBS to another metric developed for use in Pennsylvania - the Macroinvertebrate Biotic Integrity Index (MBII; Klemm et al. 2003). The large-scale analyses and careful testing of the MBII make it an ideal index for comparison with TBS. Strong similarity between the two indices would indicate that they convey much the same information, while substantial differences would indicate a need for re-evaluation of the TBS.

The MBII was developed by the U.S. EPA’s Environmental Monitoring and Assessment Program. The program evaluated 574 first, second, and third order stream reaches in a region known as the Mid-Atlantic Highlands (Klemm et al. 2003). The Mid-Atlantic Highlands include Pennsylvania, Maryland, Virginia, West Virginia, parts of Delaware outside the coastal Plains, and the Catskill Mountains of New York. Importantly, the region includes southwestern Pennsylvania and samples from southwestern Pennsylvania streams were included in the study. Klemm et al. (2003) evaluated more than 100 macroinvertebrate metrics that are commonly used in the published literature, testing signal to noise ratio, responsiveness to disturbance gradients, and redundancy (i.e. correlation) among the metrics. The resulting multi-metric MBII was intended, like the PADEP’s TBS, to indicate the extent to which disturbances have impacted a stream reach.

The University further assessed the effectiveness of PADEP’s TBS index by comparing it to the indices developed for three nearby states whose ecosystems are similar to the ecosystem in which the majority of underground bituminous coal mining occurs in PA: Maryland’s Index of Biological Integrity (IBI; Stribling et al. 1998), the Virginia Stream Condition Index (VASCI; Burton and Gerrisen 2003) and the West Virginia Stream Condition Index (WVSCI; Gerritsen et al. 2000). Finally, the University calculated the first principle component (PC1) of the aggregate set of 22 metrics used in these five indices (see Table VII-1 for a list of the metrics used in each index). Percent Tanytarsini of the Chironomidae, a metric used in the VASCI, could not be used because mine operators in Pennsylvania are only required to report Chironomidae without tribe or genus identification; this metric was therefore omitted from both the Principal Component analysis and the VASCI calculations.

Table VII-1. Comparison of PADEP Total Biological Score component metrics and of the index itself to four other indices and their component metrics: The Mid-Atlantic Highlands Macroinvertebrate Biological Integrity Index (MBII) of Klemm et al. (2003), The Maryland Index of Biological Integrity (IBI) of Stribling et al. (1998), The Virgina Stream Condition Index (SCI) of Burton and Gerrisen (2003) and the West Virgina Stream Condition Index (SCI) of Gerritsen et al. (2000). All correlations reported are based on pre-normalized metrics. The average correlations do not change significantly when normalized metrics are used.

Macroinvertebrate Metric	PADEP TBS	Mid-Atlantic Highlands MBII	Maryland IBI	Virgina SCI	West Virginia SCI	PC1
Taxonomic Richness	✔		✔	✔	✔	✔
% of Individuals in 5 Most Dominant Taxa		✔				✔
% of Individuals in 2 Most Dominant Taxa				✔	✔	✔
Trichoptera Taxa Richness	✔	✔				✔
Ephemeroptera Taxa richness		✔	✔			✔
Plecoptera Taxa Richness		✔				✔
Diptera Taxa Richness			✔			✔
Filterer-Collector + Predator Taxa Richness	✔					✔
Filterer-Collector index		✔				✔
EPT (Ephemeroptera + Trichoptera + Plecoptera) Taxa Richness			✔	✔	✔	✔
Intolerant Taxa Richness	✔		✔			✔
Macroinvertebrate Intolerance Index (MTI)		✔				✔
% (Plecoptera + Trichoptera - Hydropsychidae)				✔		✔
% EPT Richness	✔					✔
% Non-Insect Individuals		✔				✔
% Ephemeroptera			✔	✔		✔
% Chironomidae				✔	✔	✔
% Tanytarsini of Chironomidae[1]			✔			
% Tolerant Individuals			✔			✔
% Collectors			✔			✔
% Scrapers				✔		✔
HBI - Hilsenhoff Biotic Index, Family Level*				✔	✔	✔
Correlation with TBS	-	0.83	0.39	0.66	0.87	0.85
Probability of a larger r^2	-	< 0.0001	0.0003	< 0.0001	< 0.0001	< 0.0001
Mean	66.9	48.9	3.9	67.9	73.9	22.46
Standard Deviation	19.3	15.6	0.66	14.7	14.2	20.95

*Abundance-weighted avg tolerance (Family taxonomic level)

[1]Not included - see Sec. VIIB.

To compare the macroinvertebrate indices, the University calculated TBS, MBII, IBI, VASCI, WVSCI and PC1 for 90 stream samples with data in the PADEP paper files. For stream samples to be usable for this analysis, the data on individual taxa abundance had to be included. As recommended by Barbour et al. (1999), all of the metrics used for calculating each state's indices are normalized according to the regional distribution of metric values. This is done by dividing individual sampling station metrics by percentiles of the regional distributions of metric values to put all metrics on scales of 0 – 100 (except for the IBI; it is scaled 0 – 10). The University did not use the PADEP TGD percentiles for the metrics, but instead used the percentiles of the metric distribution for the 90 samples in this analysis. This was done so that all five of the indices being compared were scaled to the same distribution. Interestingly, the mean value of the TBS for the 90 samples calculated by this method (67.0 +/- 2.0 SE) fall within two standard errors, i.e. not detectably different, from the two pre-mining mean TBS the University reported in Section VII.H.2 (71.2 +/- 1.5 SE; 67.7 +/- 2.9 SE), indicating that the use of any of those distributions' percentiles will give nearly identical values. The University wrote a Statistical Analysis System (SAS 2013) macro language program (available on request from the University research team) that estimated the percentiles of the distribution of values across the 90 stream samples and used those percentiles to norm the metrics according to the methods outlined by PADEP (2005b) and the developers of the other indices cited above. The program then calculated the TBS and other indices from their respective normalized metrics. Finally, the program calculated the correlations among the five measures. The results are shown at the bottom of Table VII-1.

PADEP's TBS is significantly correlated with all five tested metrics and highly correlated with all but Maryland's IBI. In particular, the high correlation of the TBS with PC1 and MBII (r^2 = 0.85 and 0.83, respectively), the most comprehensive of the indices tested, shows that the TBS captures the vast majority of information on macroinvertebrate community integrity available from the methods commonly used across this region for bioassessment. The University is confident in stating that the PADEP TBS is an accurate indicator of changes in the biotic community associated with disturbance and pollution.

VII.C – Isolating the Effect of Mine Subsidence: Controlling for Non-Mining Related Influences on Stream Flow and Biology

Stream flow and biology are naturally variable and influenced by a number of factors unrelated to mining. Below is a discussion of the degree to which climate, catchment and reach-scale characteristics as well as sampling season influence stream flow and biology above longwall mines in Pennsylvania. The University's efforts to control for these factors while investigating subsidence impacts are also described.

VII.C.1 – Effect of Climate on Stream Flow

Flow regimes are strongly affected by many factors, including catchment size, geology, topography, surrounding vegetation, local temperatures and precipitation (Poff et al. 1997). While all of these factors influence a stream's flow pattern, it is the latter climatic factors of temperature and precipitation that drive seasonal and annual variation in stream flow. Periods of high and low stream flows are common in Pennsylvania due to the seasonal climate (Section VI). During periods of low flows, streams can even experience flow loss if the climatic conditions are dry enough. However, flow loss is also a common impact of mine subsidence. Thus, to understand the effect of subsidence on flow loss, it is critical to first understand the effect of climate on stream flow loss.

The relationship between climatic conditions and stream flow loss was assessed by first determining periods of drought during the 4th Act 54 assessment period. Drought conditions are regularly monitored by PADEP using a composite metric that combines data from four drought indicators – groundwater percentiles, surface water percentiles, precipitation departures, and the Palmer Drought Severity Index (Pennsylvania Water Science Center 2014). Using this composite metric, counties are placed into one of three drought categories: drought watch, drought warning, and drought emergency (Pennsylvania Water Science Center 2014). Greene and Washington counties, where all active longwall mines from the 4th assessment period are located, experienced three drought watches and one drought warning during the reporting period (Table VII-2).

Table VII-2. Periods of drought conditions during the 4th Act 54 assessment in areas with active longwall mines.

Start Date	End Date	Status
11/7/2008	1/26/2009	Drought watch
9/16/2010	11/10/2010	Drought warning
11/10/2010	12/17/2010	Drought watch
7/19/2012	8/31/2012	Drought watch

Next, the average percent flow loss (defined as: (the length of the stream with no flow) / (total length of the stream observed)*100) was calculated for streams (N = 18) during a non-drought (March 2010) and drought period (October 2010). The selected streams are located in the permit area for Bailey Mine, but had not yet been undermined at the time of the March and October 2010 surveys. Many of these streams exhibited substantial flow losses during the drought period (Figure VII-1). Large flow losses were particularly common on first order streams. In the non-drought period, only 5 of the 18 streams experienced flow loss, and all flow losses were minimal (<11% of stream dry; Figure VII-1). Clearly, climate must be accounted for when evaluating the effect of mine subsidence on stream flow loss.

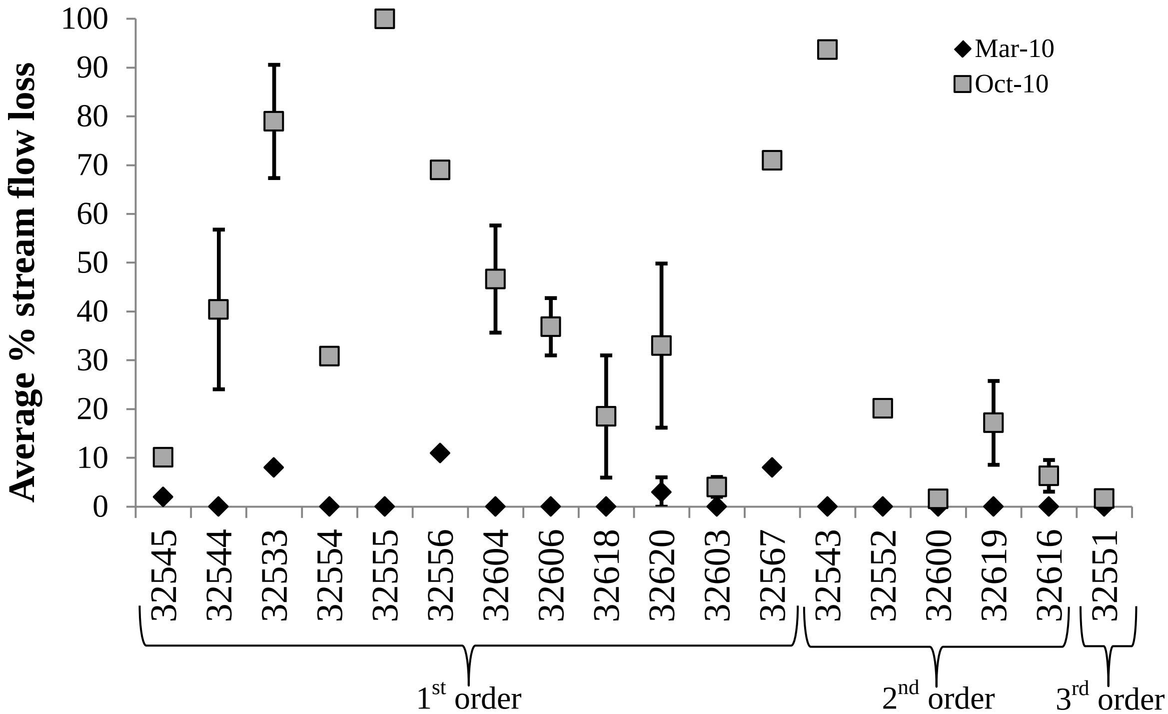

Figure VII-1. Average percent stream flow loss on streams outside of mining under non-drought (March 2010) and drought conditions (October 2010). Data are means +/- one standard error. Streams without error bars were either monitored only one week out of the month or exhibited no variation from one week to another.

To that end, mine operators with approval from PADEP recently established a "wet" and "dry" season for Pennsylvania (Consol Pennsylvania Coal Company 2010). The wet season is December-May, while the dry season is June-November. The University adopted this classification system in the assessment of stream flow loss to avoid confounding the impact of climatic conditions with mine subsidence. Below, maximum and minimum stream flow losses are reported for both the wet and dry seasons (Section VII-E; Appendix F).

VII.C.2 – Effect of Catchment and Reach-Scale Characteristics on Stream Biology

In stream ecosystems, the distribution and abundance of macroinvertebrates is determined by a spatial hierarchy of habitat filters (sensu Poff 1997; Vinson and Hawkins 1998, Heino 2009). The abiotic and biotic filters range from broad-scale controls at the stream catchment basin to local factors operating within the stream reach. At the catchment scale, land use is one factor that is known to influence macroinvertebrate communities (reviewed by Johnson and Gage 1997, Allan 2004, Hughes et al. 2006, Johnson and Host 2010). The conversion of land to agricultural and developed areas impacts stream communities by changing both the physical and chemical makeup of aquatic environments (Allan 2004). Specifically, human dominated landscapes can increase stream sedimentation rates, introduce contaminants, alter runoff and groundwater flows, and reduce the amount of riparian vegetation and large woody debris (Allan 2004). These physiochemical changes scale up and affect the diversity of life streams are able to support. The loss of riparian vegetation can reduce detrital food inputs for macroinvertebrates while the loss of coarse woody debris can eliminate refuges from predation for fish and invertebrates (Everett and Ruiz 1993). When evaluating the magnitude of land use impacts, it is important to consider the spatial arrangement of land use in a watershed (Kearns et al. 2005). For example, a large agricultural field adjacent to a stream is likely to exert very different effects on the stream ecosystem relative to several small fields scattered throughout an otherwise forested watershed. While land use patterns in the catchment can influence macroinvertebrate diversity, the relative importance of land use vs. factors operating at the reach-scale remains the subject of much investigation (Johnson and Host 2010). Reach-scale factors include channel morphology, stream habitat, and water quality. Clearly, the environmental conditions at the reach-scale are regulated to some degree by factors operating at the catchment scale (Poff 1997). However, in some areas – particularly those that are largely undisturbed – reach-scale factors can outweigh the importance of catchment factors and play a dominant role in structuring macroinvertebrate communities (Weigel et al. 2003, Johnson et al. 2007)

The complex nature of interactions between stream ecosystems and their surrounding landscapes required that the University develop a method to control for this complexity before assessing the impacts of mine subsidence (Section VII.H) and restoration on stream macroinvertebrate communities (Section VII.J). Land cover information was obtained from the National Land Cover Database (NLCD) 2006 Land Cover data layer (Fry et al. 2011) for 16 stream catchments over areas of active and planned longwall mining (Bailey, Enlow Fork, Cumberland, and Emerald Mines). Within each catchment, subwatersheds were delineated for each stream bio-monitoring station (N = 201 stations; Figure VII-2) using a flow accumulation layer and the "Watershed" tool in ArcGIS. The NLCD layer was clipped to the resulting subwatersheds and then FragStats version 4.1 (University of Massachusetts, Amherst, Massachusetts, USA) was used to calculate land use and landscape pattern metrics (i.e. largest patch index, edge density, shape index, contiguity, patch richness, Simpson's landscape diversity index, Simpson's landscape evenness; see McGarigal et al. 2012 for definitions). Lastly, the landscape data was merged with information on reach-scale characteristics for each of the bio-monitoring stations. Reach-scale data were collected by the mine operator prior to undermining the stream. Measures of water quality included conductivity, pH, and dissolved oxygen. Measures of stream habitat and channel morphology were based on the U.S. Environmental Protection Agency low-gradient habitat assessment score for each station. The assessment evaluates 10 habitat parameters, such as channel sinuosity, bank stability, and riparian zone vegetation, to generate a composite score. Scores range from 0-200, with scores in the 156-200 range representing optimal habitat, 106-155 suboptimal, 56-105 marginal, and 0-55 indicating poor habitat.

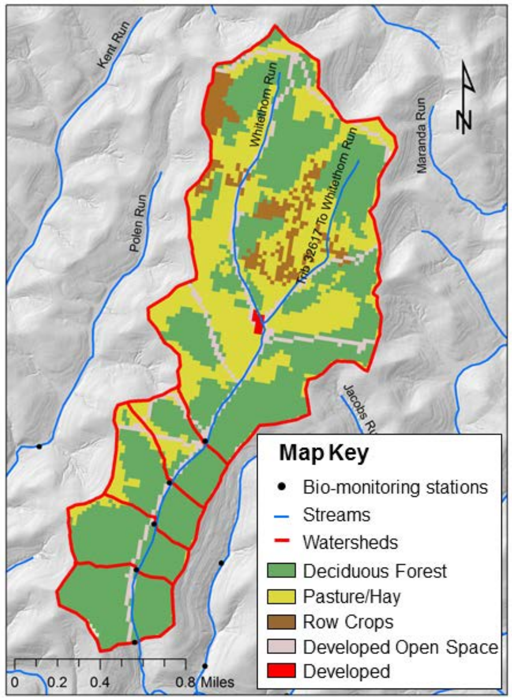

Figure VII-2. An example of land use within Whitethorn Run, a focal catchment in a planned expansion area of Bailey Mine. The red lines indicate the delineated subwatersheds for each bio-monitoring station (black point) within the catchment. The subwatersheds in this catchment are dominated by forest and pasture/hay land use types.

Data from large spatial scales can be challenging to statistically analyze for several reasons (Allan and Johnson 1997, King et al. 2005). One problem is that stream sites that are close together are likely to be more similar in terms of land use and biology than sites that are distant from each other (i.e. spatial autocorrelation; King et al. 2005). Due to the close proximity and sharing of water flows between many of the bio-monitoring stations in these 16 catchments, spatial autocorrelation was assessed prior to further analysis. Using ArcGIS, the stream course distance between stations that were hydrologically connected (a "station pair") was measured. Stations were considered to be connected if water flowed downstream from one site to another (i.e. "flow-connected", sensu Peterson and Ver Hoef 2010). This method does not consider stations from adjacent tributaries to be "connected". This approach is appropriate for sites in Appalachia because lateral dispersal of macroinvertebrates between adjacent watersheds via adult flights is rare (Griffith et al. 1998). Based on the distribution of the distances between stations, the station pairs were organized into 10 bins and the average correlation in Total Biological Score was calculated for each bin (N = 29-95 station pairs/ bin). A bootstrapping procedure was used to generate 95% confidence intervals for each correlation estimate. The analysis indicated that stations within 1,673-ft of each other had significantly correlated Total Biological Scores. To eliminate spatially autocorrelated stations from the data set, all upstream stations that were hydrologically connected to the most downstream station and within the determined distance were eliminated. This procedure was applied to each station until the entire watershed had been examined. This analysis removed 50 stations from the data set, leaving 151 stations that can be considered statistically independent of each other.

A second statistical problem is that catchment and reach-scale variables are often not independent predictors of biological communities because they can be correlated with each other. For this data set, a Spearman correlation analysis revealed significant correlations among many of the catchment-scale variables (Appendix E). While land use was also expected to be highly correlated with water quality, there were few highly significant relationships (Appendix E). The lack of strong correlations suggests that stream pH, conductivity, and dissolved oxygen in southwestern Pennsylvania may be affected to a larger degree by the area's underlying geology than land use practices. Prior to further analysis, bio-monitoring stations that were missing data for one or more of the predictor variables were dropped (7 stations dropped; N = 144).

A partial least squares (PLS) regression was used to analyze the relationship between Total Biological Score and the catchment and reach-scale variables. PLS regression is a multivariate statistical technique that is well-suited for ecological datasets with correlated predictor variables (Carrascal et al. 2009). PLS regression extracts a series of orthogonal factors from the predictor variables. The factors are extracted in such a way as to maximize the variance explained among the predictors and in the response variable (SAS Institute Inc. 2008). While this technique initially extracts as many factors as there are predictors in the dataset (in this case, 17 factors), the number of relevant factors can be determined by looking at the explanatory power of each factor. In the analysis, the first two factors explained 32.8% of the variation in Total Biological Score (Table VII-3). The remaining factors only explained an additional 3.1% of the variance in Total Biological Score, suggesting that the bulk of the explanatory power is contained within these first two factors. The meaning of the two factors can be determined by evaluating the weights of each predictor variable on the factors (Table VII-3). For factor 1, negative values are associated with bio-monitoring stations that have diverse catchments with pasture/hay and reaches with high pH and low habitat scores. In contrast, stations with positive values of factor 1 have catchments that are dominated by large, intact patches of deciduous forest and reaches with neutral pH and high habitat scores. Like factor 1, factor 2 is largely a function of a station's habitat assessment scores. However, factor 2 is also a function of the % developed land in the catchment and landscape contiguity. Bio-monitoring stations with negative values of factor 2 generally have contiguous landscapes in their catchment but low habitat scores at the reach scale. Stations with positive values of factor 2 have catchments that are somewhat fragmented by development and reaches with high habitat scores.

Table VII-3. Weights for each of the catchment and reach-scale variables on factors 1 and 2 from the partial least squares regression. The four largest weights for each factor are in bold.

Predictor	**Weights**	
Catchment scale	**Factor 1**	**Factor 2**
% Pasture	**-0.397**	0.016
% Deciduous Forest	0.360	-0.184
% Developed Open Space	-0.045	0.342
% Row Crops	-0.120	0.189
% Developed	0.145	**0.428**
Largest Patch Index	**0.382**	-0.135
Edge Density	-0.265	**0.348**
Shape Index (AM)	-0.249	-0.132
Contiguity (AM)	0.130	**-0.379**
Patch Richness	-0.263	-0.049
Simpson's Diversity Index	-0.327	0.277
Simpson's Evenness Index	-0.301	0.266
Watershed Area	-0.171	-0.128
Reach scale		
US EPA Low-Gradient Habitat Assessment	**0.465**	**0.483**
pH	**-0.396**	-0.281
Conductivity	-0.182	-0.019
Dissolved oxygen	-0.044	-0.130
R^2 for Y-variable	0.189	0.140
R^2 for X-variables	0.334	0.128

Using Spearman correlation analysis, the University found that both factors from the PLS regression have a significant positive correlation with TBS (factor 1: $r_s = 0.47$, $P < 0.0001$; factor 2: $r_s = 0.34$, $P < 0.0001$; Figure VII-3). The positive correlation between TBS and factor 1 (Figure VII-3a) indicates that larger TBS are associated with forested catchments and reaches with neutral pH and pristine habitat. Low TBS are associated with pasture/hay land use in the catchment, alkaline pH, and poor reach habitat. The University is aware of only one other study that has assessed the relationship between agricultural land use and stream biology in Pennsylvania (Genito et al. 2002). The study found that Ephemeroptera, Trichoptera, and overall taxonomic richness decreased as agriculture increased in a subwatershed of a tributary to the Susquehanna River (Genito et al. 2002). The University's results generally confirm those of Genito et al. (2002) and indicate that agricultural land use reduces the quality of macroinvertebrate communities in Pennsylvania.

The positive correlation between TBS and factor 2 (Figure VII-3b) indicates that larger TBS are also associated with catchments with developed land and reaches with high habitat scores. Development and/or urbanization is generally associated with declines in macroinvertebrate community metrics (e.g. Roy et al. 2003, Urban et al. 2006). However, Allan (2004) notes that declines are often only apparent once a threshold level of development or urbanization (~10-20%) has been reached. In the University's dataset, the average percent developed land in a catchment was 0.25% and the maximum percent developed was 7.1%. Thus, it is likely that the degree of development in this region is not large enough to significantly impair macroinvertebrate community health. In fact, the slight increases in light, nutrients, and water temperature that accompany small amounts of development may actually result in increases in apparent stream biotic integrity up to a point.

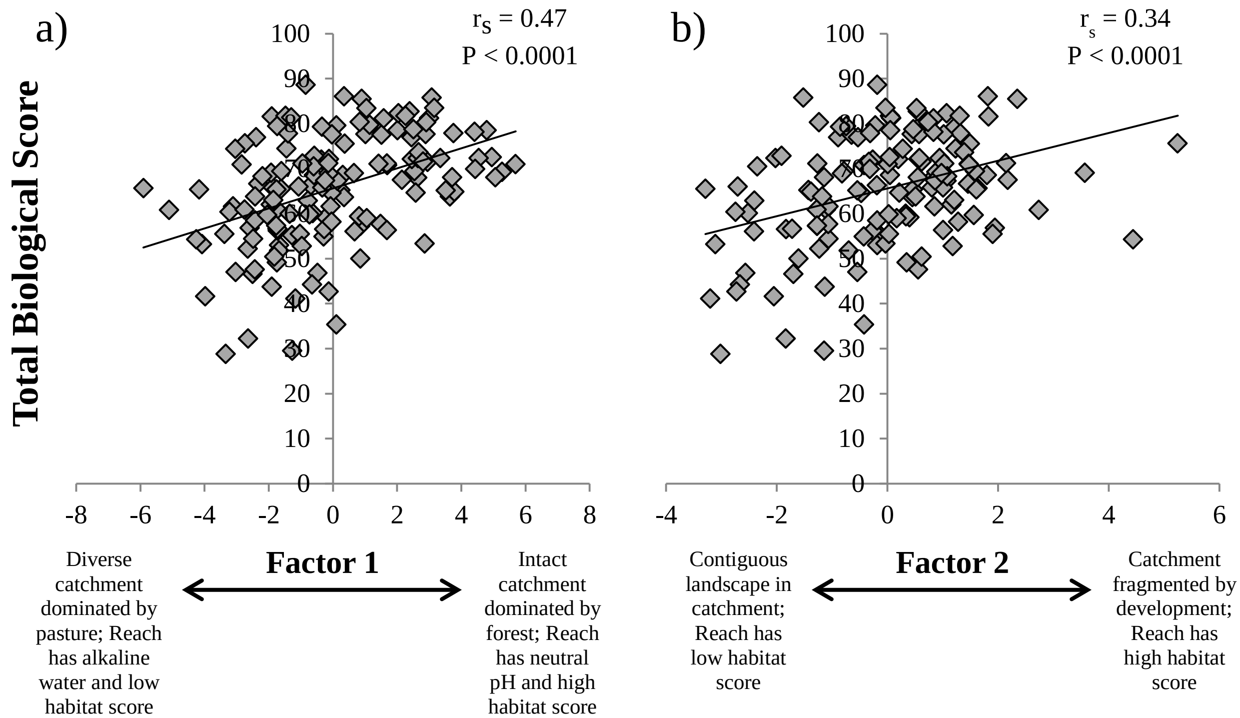

Figure VII-3. Correlation between Total Biological Score and factors 1 and 2 from the PLS regression (N = 144 bio-monitoring stations).

Lastly, the variable importance for projection (VIP) value was calculated for each predictor variable. The VIP value indicates the importance of a variable for fitting the overall PLS regression model (SAS Institute Inc. 2008). Variables with VIP values less than 0.8 are generally unimportant for model fitting and prediction (SAS Institute Inc. 2008). In this analysis, two reach-scale predictors – the U.S. EPA low gradient habitat assessment score and stream pH – were the most important variables for modelling TBS (Table VII-4). Descriptions of landscape fragmentation (i.e. Simpson's diversity index and edge density) and the % developed land in the catchment were also highly important.

The results of this analysis were used by the University to control for the effect of land use while investigating subsidence impacts on stream biology (Section VII.H). For each station, the scores for factors 1 and 2 were multiplied by the regression slope for TBS v. factor 1 or TBS v. factor 2. This product was then subtracted from the station's TBS. The resulting "adjusted TBS" puts all streams on a level playing field during assessment of mining impacts (e.g. TBS from forested sites are adjusted downwards while TBS from agricultural sites are adjusted upwards) and allows for biological comparisons across sites.

Table VII-4. Variable importance for projection (VIP) values for each predictor variable in the PLS regression model. Values greater than 0.8 are in bold and indicate predictors that are important for describing TBS.

Predictor	**VIP Value**
Catchment scale	
% Pasture	**1.079**
% Deciduous Forest	**1.080**
% Developed Open Space	**0.861**
% Row Crops	0.573
% Developed	**1.135**
Largest Patch Index	**1.092**
Edge Density	**1.126**
Shape Index (AM)	0.751
Contiguity (AM)	**1.008**
Patch Richness	0.724
Simpson's Diversity Index	**1.125**
Simpson's Evenness Index	**1.053**
Watershed Area	0.564
Reach scale	
US EPA Low-Gradient Habitat Assessment	**1.744**
pH	**1.283**
Conductivity	0.496
Dissolved oxygen	0.345

VII.C.3 – Effect of Month of Sampling on Stream Biology

Temporal variation in stream macroinvertebrate communities can result from differences in macroinvertebrate life histories as well as seasonal and annual changes in habitat availability, temperature, and flow (Linke et al. 1999). One particularly strong example of seasonal variation in macroinvertebrate abundance comes from the family Capniidae (Plecoptera). Due to their life cycle, Capniidae larvae are very abundant in Pennsylvania streams across the winter months, but are rare in the summer months (Walsh et al. 2007), earning them the common name of "winter stoneflies". Summer and winter months are known to consistently vary in terms of taxa richness and diversity (e.g. Linke et al. 1999, Riley et al. 2007). As a result, many bio-monitoring programs select an index period, or a temporal constraint, for sampling (e.g. West Virginia Stream Condition Index, Gerritsen et al. 2000; Virginia Stream Condition Index, Burton and Gerritsen 2003). In Pennsylvania, TGD 563-2000-655 requires that all Total Biological Scores (TBS) be collected between October and May (PADEP 2005a). However, even within an index period, biological metrics can vary with time (Gerritsen et al. 2000).

To determine the degree of both seasonal and annual variation in TBS, the University analyzed data on stream TBS and sampling date that were collected prior to mining by mine operators (N = 1,328 samples). Analysis of variance (ANOVA) was used to test the following model: TBS = month + year + station. Because multiple samples were collected at a single station, station was included to account for the sample variation that was due to site level differences. Overall, the ANOVA model was highly significant ($P < 0.0001$) and accounted for a significant amount of the variation in TBS (R^2 = 0.72). TBS varies significantly with month of sampling across the October-May index period (month, $F_{7,805}$= 5.30; P < 0.0001). Samples collected during the beginning and end of the index period tend to have lower TBS than samples collected in the middle of the period (Figure VII-4). Samples collected in October and November have particularly low TBS, scoring on average 10-11 points lower than samples collected in December-March. TBS for samples collected in April and May are also lower than those from December-March, although to a lesser degree (3-4 points lower on average). May scores are actually not significantly different from those collected in December-February (Figure VII-4). The observed variation in TBS across the index period may be a function of macroinvertebrate life history events (Figure VII-4), although additional studies are needed to test this idea. Interestingly, year was not a significant predictor of TBS ($F_{6,805}$= 1.42; P = 0.20) suggesting that year-to-year fluctuations in abiotic conditions play little to no role in explaining macroinvertebrate community variation.

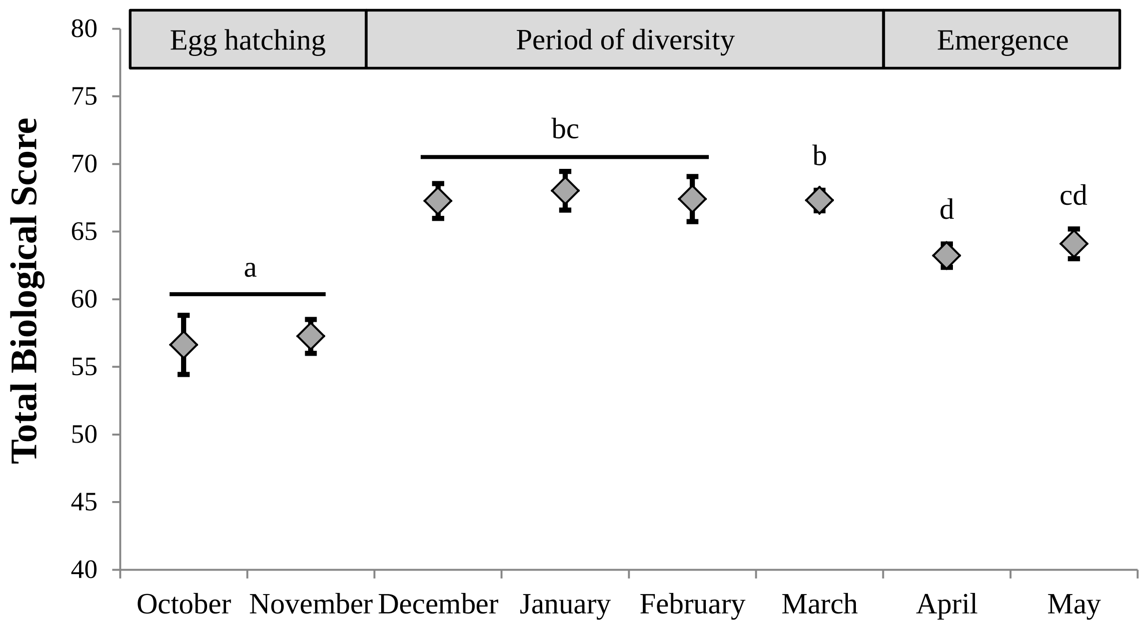

Figure VII-4. Relationship between Total Biological Score and month of sampling across PADEP's sampling index period (N = 1,328 samples). Data are monthly mean TBS +/- 1 standard error. Points that share a letter are not significantly different from each other (P > 0.05, Tukey-Kramer HSD test). Gray boxes represent possible biological mechanisms underlying differences in scores over time.

An ideal index period will balance the need to minimize temporal variability with the goal of maximizing sampling of targeted assemblages (Barbour et al. 1999). Due to the strong reduction in average TBS that occurs in October and November sampling events, the University suggests that PADEP shorten its index period for TBS sampling to December-May. Greater than 50% of the pre-mining samples in this dataset were collected in March and April, suggesting that mine operators are already focusing their sampling efforts in the early spring months.

To control for temporal variation while analyzing the effect of subsidence on stream biology (Section VII.H), each station's TBS was adjusted based on the month in which it was collected. A standard adjustment value was created for each month by subtracting the average TBS for each month from the overall average TBS. This correction value was then subtracted from each bio-monitoring station's TBS. As with the land use adjustment described above, the goal was to equalize all samples in terms on sampling month.

VII.D – Length of Streams Undermined During the 4th Assessment

PADEP tasked the University with determining the miles of streams undermined during the 4th Act 54 assessment period. To accomplish this task, ArcGIS was used to clip the "Networked Streams of PA" (see Section II.B.1 for description) to the longwall, room-and-pillar, and pillar recovery extents (where applicable). The advantage to using the "Networked Streams of PA" in this analysis is that it was used by two previous Act 54 reports (Conte and Moses 2005, Iannacchione et al. 2011), so results are directly comparable with those reports. The drawback to using this streams layer is that it does not include many of the smaller tributaries in Pennsylvania (hereafter referred to as zero-order tributaries). As a result, the analysis may underestimate the actual miles of stream undermined. The streams layer was also clipped to the 200-ft buffer surrounding each mine, as streams in this area are considered by PADEP to be potentially impacted by mining.

A total of 96.05 miles of streams were undermined during the 4th assessment period (Table VII-5). Of these, 50.59 miles of streams were undermined by longwall mining methods, while 45.04 miles were undermined by room-and-pillar methods (Table VII-5). Less than 0.5% of the streams undermined during the assessment period were undermined by pillar recovery techniques (Table VII-5). Overall, the 4th assessment period experienced a 16% decrease in stream lengths undermined relative to the 3rd Act 54 assessment period, during which a total of 113.7 miles of streams were undermined (Iannacchione et al. 2011). Miles of undermined stream declined in both the longwall and room-and-pillar categories (longwall – 3rd assessment: 63.81; room-and-pillar – 3rd assessment: 49.7; Iannacchione et al. 2011).

Table VII-5. Lengths of streams undermined in miles by mine and mining method.

Mine	Mining Method				Total w/o Buffer (mi)	Total Length (mi)
	Room-and-Pillar	Long Wall	Pillar Recovery	Buffer Zone		
	Length (mi)	Length (mi)	Length (mi)	Length (mi)		
4 West	3.12	0.00	0.43	2.89	**3.55**	**6.44**
Agustus	0.77	0.00	0.00	0.38	**0.77**	**1.15**
Bailey	2.86	14.36	0.00	3.37	**17.22**	**20.59**
Barrett Deep	0.11	0.00	0.00	0.20	**0.11**	**0.31**
Beaver Valley	0.22	0.00	0.00	0.38	**0.22**	**0.61**
Blacksville 2	2.56	5.81	0.00	1.91	**8.37**	**10.28**
BMX	2.46	0.00	0.00	2.40	**2.46**	**4.86**
Cherry Tree	2.05	0.00	0.00	2.07	**2.05**	**4.12**
Clementine 1	1.19	0.00	0.00	1.79	**1.19**	**2.98**
Crawdad	0.37	0.00	0.00	0.38	**0.37**	**0.75**
Cumberland	2.37	7.62	0.00	2.88	**9.99**	**12.86**
Darmac 2	0.81	0.00	0.00	1.88	**0.81**	**2.69**
Dora 8	0.41	0.00	0.00	0.59	**0.41**	**1.00**

Mine	Mining Method				Total w/o Buffer (mi)	Total Length (mi)
	Room-and-Pillar	Long Wall	Pillar Recovery	Buffer Zone		
	Length (mi)	Length (mi)	Length (mi)	Length (mi)		
Dutch Run	0.48	0.00	0.00	0.58	**0.48**	**1.06**
Emerald	2.16	6.04	0.00	4.38	**8.21**	**12.59**
Enlow Fork	5.00	16.59	0.00	4.23	**21.59**	**25.82**
Geronimo	0.00	0.00	0.00	0.03	**0.00**	**0.03**
Gillhouser Run	0.65	0.00	0.00	0.39	**0.65**	**1.04**
Harmony	0.20	0.00	0.00	0.22	**0.20**	**0.42**
Heilwood	0.67	0.00	0.00	1.19	**0.67**	**1.86**
Horning Deep	0.05	0.00	0.00	0.10	**0.05**	**0.15**
Kimberly Run	0.83	0.00	0.00	0.75	**0.83**	**1.58**
Knob Creek	0.51	0.00	0.00	0.36	**0.51**	**0.86**
Little Toby	0.09	0.00	0.00	0.52	**0.09**	**0.61**
Logansport	2.75	0.00	0.00	3.17	**2.75**	**5.91**
Long Run	0.03	0.00	0.00	0.04	**0.03**	**0.08**
Lowry Deep	0.21	0.00	0.00	0.53	**0.21**	**0.73**
Madison	0.83	0.00	0.00	0.16	**0.83**	**0.99**
Miller Deep	0.19	0.00	0.00	0.08	**0.19**	**0.27**
Mine 84	0.08	0.17	0.00	0.11	**0.25**	**0.36**
Nolo	0.28	0.00	0.00	0.38	**0.28**	**0.67**
Ondo	0.25	0.00	0.00	0.89	**0.25**	**1.14**
Prime 1	0.05	0.00	0.00	0.58	**0.05**	**0.64**
Penfield	0.33	0.00	0.00	0.33	**0.33**	**0.66**
Quecreek 1	2.31	0.00	0.00	1.64	**2.31**	**3.95**
Rossmoyne 1	0.13	0.00	0.00	0.55	**0.13**	**0.68**
Roytown	0.89	0.00	0.00	0.58	**0.89**	**1.47**
Sarah	0.22	0.00	0.00	0.67	**0.22**	**0.89**
Starford	0.16	0.00	0.00	0.48	**0.16**	**0.64**
Titus Deep	0.22	0.00	0.00	0.17	**0.22**	**0.39**
TJS 5	0.00	0.00	0.00	0.07	**0.00**	**0.07**
TJS 6	0.98	0.00	0.00	1.20	**0.98**	**2.19**
Toms Run	1.57	0.00	0.00	1.70	**1.57**	**3.27**
Tracy Lynne	1.02	0.00	0.00	1.67	**1.02**	**2.68**
Twin Rocks	0.58	0.00	0.00	0.60	**0.58**	**1.18**
Windber 78	2.01	0.00	0.00	1.54	**2.01**	**3.55**
	45.04	**50.59**	**0.43**	**51.01**	**96.05**	**147.06**

Using data from the Pennsylvania Aquatic Community Classification System, it was determined that many of the streams that were undermined are located in watersheds of high conservation concern for this region of Pennsylvania (Table VII-6; Walsh et al. 2007). Streams in these watersheds meet specific criteria for high quality community habitats, including macroinvertebrate diversity and trophic structure, fish metrics and the number of stream reaches designated as “Least Disturbed Streams” (i.e. having little to no human influence; Walsh et al. 2007). Scientists with the Pennsylvania Natural Heritage Program and the Western Pennsylvania Conservancy recommend that “…monitoring agencies and programs…scale their efforts and funding for conservation and restoration based on the regional prioritization” (Walsh et al. 2007).

Protection of streams in these high quality watersheds is thus an important conservation objective.

*Table VII-6. Watersheds that were undermined during the 4th assessment and are classified as high conservation priorities in the Waynesburg Hills region (Walsh et al. 2007). * = Tier 1 – highest quality watersheds in the region, Tier 2 - second highest quality watersheds.*

Mine	Watershed Name	Conservation Priority Tier*
Bailey	Dunkard Fork	1
Bailey	South Fork of Dunkard Fork	1
Cumberland	Whiteley Creek	2
Emerald	Whiteley Creek	2
Emerald	South Fork of Tenmile Creek	2
Enlow	Templeton Fork	2

VII.E – Length of Streams Impacted During the 4th Assessment

In addition to determining the total miles of streams undermined, PADEP also requested that the University quantify the length of stream impacts. Specifically, the University was tasked with the following:

> **Task 1.** Calculate the lengths of undermined streams (organized by mining method) that fall into one of the following categories – a) streams with no reported effects, b) streams with mining-induced pooling, and c) streams with mining-induced flow loss.
> **Task 2.** Identify the location of each reported incident of stream flow loss. Provide a GIS layer of each mine showing the location of all reported incidents of flow loss that were longer than two weeks or required augmentation and a table that lists the latitude and longitude of the center of the flow loss and the minimum and maximum lengths.
> **Task 3.** Identify the location of each reported incident of pooling. Provide a GIS layer of each mine showing the location of all reported incidents of stream pooling and a table that lists the latitude and longitude of the center of the pooling and minimum and maximum lengths.

To complete these tasks, all undermined streams had to be classified as unaffected, impacted by mining-induced flow loss, impacted by mining-induced pooling, or impacted by both flow loss and pooling. To categorize the undermined streams, the University used data from PADEP monthly stream flow map database and the mine permit revisions. The monthly stream flow map database is a recent addition to the data files at CDMO, so it is briefly described here. The database is stored in a Microsoft Outlook folder on PADEP servers. Each month, mine operators submit stream hydrologic assessments to the CDMO database via email. Because there is no standardized format for these data submissions, stream flow maps and data files from different mine operators vary in content. For mines operated by Consol Energy, Inc., maps and Microsoft Excel files identify the location and length of flow loss on a weekly basis. The Excel files also report which streams received augmentation each week, including the number of active augmentation wells per stream and the augmentation rate. Maps identify the location of all augmentation wells. For mines operated by Alpha Natural Resources, Inc., maps identify the location of flow losses on a weekly basis, but there are no accompanying Excel files that quantify the length of flow loss or that describe patterns of augmentation. Because the files from Alpha Natural Resources, Inc. did not include data on stream augmentation, the University asked CDMO to request these data from the mine operator. CDMO requested the data and additional maps were supplied to the University on a cd. The maps that were received are known as "straight line" maps. The stream of interest is depicted as a straight line with sections of flow loss/pooling and dates of augmentation color-coded (Figure VII-5).

Streams were classified using the above data. Streams that received augmentation during the 4th assessment period (see Table VII-13) were classified as experiencing mining-induced flow loss. While the contract also asked the University to consider flow losses that lasted longer than two weeks, many intermittent streams commonly lose flow for two weeks or more during the dry season, especially during drought periods (Figure VII-1). Thus, it seems inappropriate to attribute flow losses of two weeks or more to mining-induced impacts. It should be noted that augmentation is likely not a perfect indicator of mining-induced impacts either. Mine operators sometimes augment streams during the dry season to demonstrate good stewardship of the land that they have mined (J. Silvis, Consol Energy, Inc., pers. comm.). For pooling, all streams that received a gate cut (see Table VII-11) or have a proposed future gate cut (Table VII-12) were classified as experiencing mining-induced pooling. Streams receiving both augmentation and gate cuts were considered to have had both mining-induced flow loss and pooling. Because streams from room-and-pillar mines did not receive augmentation or gate cuts, this analysis focuses exclusively on streams from longwall mines.

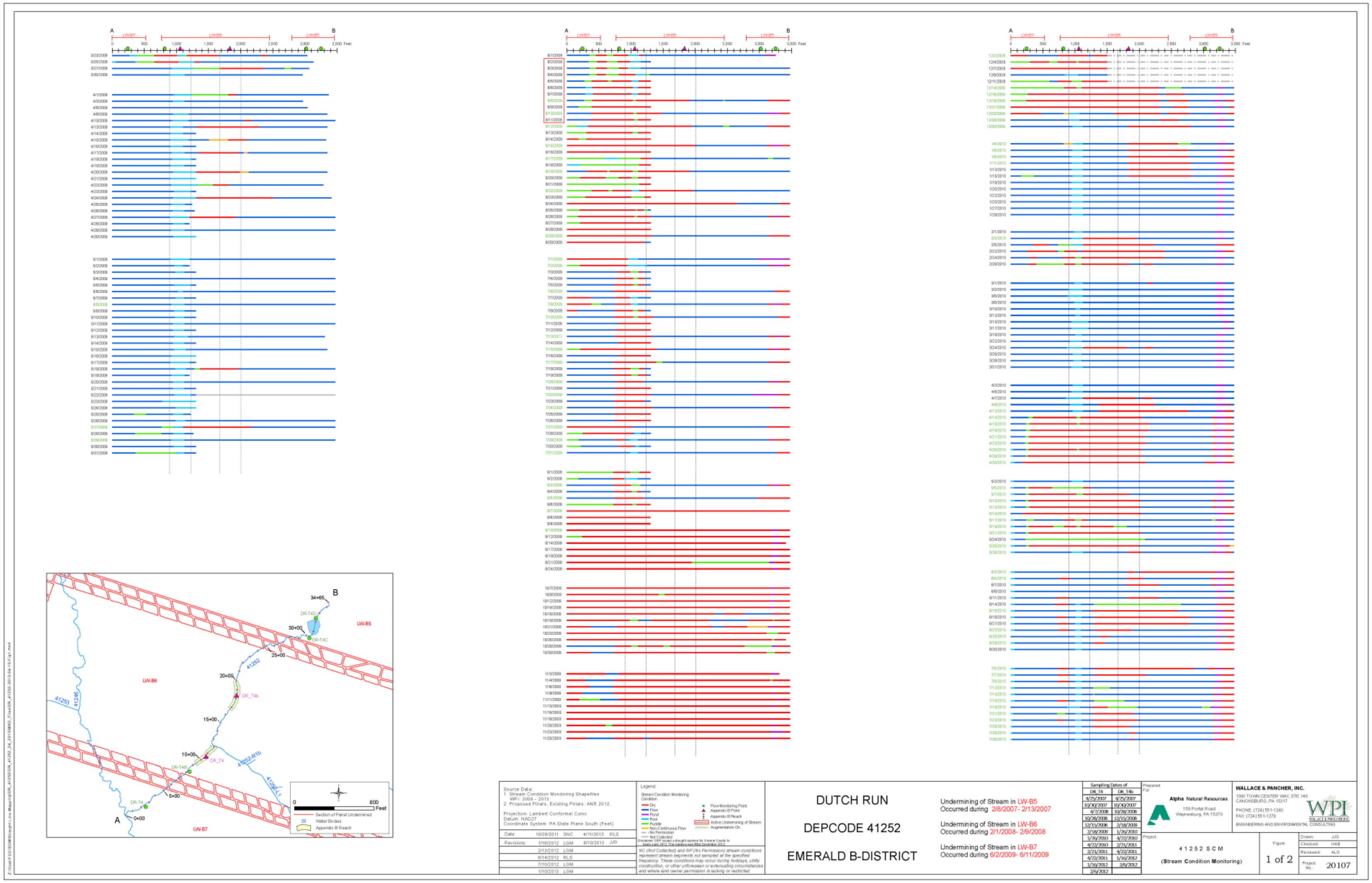

Figure VII-5. An example of a "straight-line" flow map from Alpha Natural Resources, Inc. Each line represents the dry and flowing lengths on a particular date for unnamed tributary 41252 to Dutch Run in Emerald Mine. Dates with augmentation are highlighted in green.

For Task 1, ArcGIS was used to clip the undermined streams in each category to the longwall, room-and-pillar, and 200-ft buffer extents. The “Networked Streams of PA” layer was used for this task. It is important to note that the results of Task 1 do not represent the length of stream *actually* experiencing flow loss or pooling. The lengths reported represent the miles of undermined stream belonging to a stream channel that experienced flow loss, pooling, or both somewhere along its extent.

Streams experiencing flow loss, pooling or both comprised 39.2 miles (Table VII-7) – or roughly 77% – of the total miles of streams undermined by longwall techniques (50.59 miles, Tables VII-5). Thus, only 23% of the total miles undermined by longwall techniques belonged to streams that did not experience mining-induced flow-loss or pooling. In contrast, streams experiencing flow loss, pooling, or both comprised just 44% (6.55 miles; Table VII-7) of the total miles of stream undermined by room-and-pillar techniques (14.95 miles from the five longwall mines, Table VII-5). For the stream lengths located in the 200-ft buffer zone, these lengths were most likely to be associated with streams that were unaffected by mining (10.82 unaffected miles, Table VII-7, of the total 16.77 buffer miles for the five longwall mines, Table VII-5). Overall, these data indicate that streams that are undermined by longwall mining techniques have a high probability of being impacted by either flow loss or pooling. There is variation across mines, with streams in Cumberland and Emerald Mines each having < 1 mile of their undermined streams belonging to streams impacted by flow loss (Table VII-7). This result could be an artifact of how the University categorized streams using data from PADEP. Recall that streams were categorized as experiencing mining-induced flow loss only if they received augmentation. Differences in mitigation approaches or even reporting practices across mine operators could account for the variation in miles of streams experiencing flow loss. Alternatively, the variation could represent natural differences in the geologic and hydrologic conditions between mines.

For Task 2, data from the PADEP monthly flow map database was used to identify the locations of maximum and minimum flow losses in both the wet and dry seasons. Due to the differences in reporting by Consol Energy, Inc. and Alpha Natural Resources, Inc., different approaches were adopted to complete this task for each operator.

Table VII-7. Undermined stream lengths categorized as belonging to streams with mining-induced flow loss, mining-induced pooling, mining-induced flow loss and pooling, or unaffected. Lengths do not represent length of impact but lengths of undermined stream segments that contain a reported impact.

	Flow Loss			**Pooling**			**Flow Loss and Pooling**			**Unaffected**		
Mine	**Longwall Length (mi)**	**Room-and-Pillar Length (mi)**	**Buffer Length (mi)**	**Longwall Length (mi)**	**Room-and-Pillar Length (mi)**	**Buffer Length (mi)**	**Longwall Length (mi)**	**Room-and-Pillar Length (mi)**	**Buffer Length (mi)**	**Longwall Length (mi)**	**Room-and-Pillar Length (mi)**	**Buffer Length (mi)**
Bailey	8.01	0.76	1.18	0.00	0.00	0.00	5.62	0.68	0.60	0.74	1.42	1.60
Blacksville	2.56	0.85	0.35	0.73	0.23	0.21	2.34	0.60	0.21	0.18	0.89	1.14
Cumberland	0.74	0.09	0.19	2.34	0.42	0.84	0.00	0.00	0.00	4.54	1.85	1.85
Emerald	0.09	0.00	0.04	1.39	0.00	0.25	0.77	0.09	0.09	3.80	2.08	4.00
Enlow Fork	8.29	1.83	0.82	2.08	0.38	0.46	4.25	0.63	0.71	1.97	2.16	2.24
TOTAL	**19.68**	**3.52**	**2.59**	**6.54**	**1.03**	**1.76**	**12.97**	**2.00**	**1.60**	**11.23**	**8.40**	**10.82**

For Consol-operated mines, Microsoft Excel files from the monthly flow map database were used to tabulate the flow loss lengths for all streams receiving augmentation during the 4th assessment period. It should be noted that Excel files for Blacksville 2 Mine were only submitted beginning in July 2011 so files for the beginning of the assessment period were not available. Statistical software was used to identify the dates of maximum and minimum post-mining flow loss in both the wet and dry seasons (SAS Institute Inc. 2013). For minimum flow losses, dates where the stream experienced 0-ft flow loss were excluded because 0-ft of flow loss cannot technically be considered a loss. Flow loss locations were identified by geo-referencing all monthly flow maps and digitizing the maximum and minimum flow loss areas. The monthly stream flow maps submitted by the mine operator do not use the “Networked Streams of PA” stream layer. Instead, these maps use a detailed stream layer that shows the locations of the smaller, zero-order tributaries that are not identified on the “Networked Streams of PA” layer. To accurately display both the stream and the flow loss segments reported by the mine operator, the University traced the mine operator’s stream layer in ArcGIS and used it to map maximum and minimum flow loss.

For mines operated by Alpha Natural Resources, the “straight-line” maps for all augmented streams were visually inspected to identify the dates of maximum and minimum flow loss for both the wet and dry seasons. The lengths of the flow loss segments were quantified using the Analysis feature of Adobe Photoshop (Adobe Photoshop CS5, Adobe Systems Incorporated, San Jose, CA). Using the scale bar provided on each map, the measurement scale was customized to convert from pixel lengths to length in feet. The flow loss lengths were then recorded using the Ruler tool. To identify the flow loss locations, the flow loss lengths were carefully matched to the streams layer provided by Alpha Natural Resources in ArcGIS. Like the stream layers utilized by Consol-operated mines, the Alpha streams layer is more detailed than the “Networked Streams of PA” stream layer and displays zero-order tributaries. Flow losses on two streams in Cumberland Mine (unnamed tributaries 41264 and 41267 to Dyers Fork) could not be mapped because these streams did not appear on the Alpha streams layer as they were undermined during the 3rd assessment period.

Maximum post-mining flow loss lengths in the dry season ranged from 936-ft to 10,883-ft (Appendix F). Summing the maximum flow loss lengths in the dry season for all streams indicates that maximum flow losses totaled 52.2 miles of undermined streams (Appendix F). Because the dry season flow loss lengths are influenced to a large degree by climatic conditions (Section VII-C.1), the maximum flow loss length in the wet season is likely a more precise indicator of mining-induced flow loss impacts. Maximum post-mining flow loss lengths in the wet season ranged from 96-ft to 8,106-ft (Appendix F). Maximum flow losses summed across all streams for the wet season totaled 23.7 miles of undermined streams (Appendix F). Maps identifying the locations of all maximum and minimum flow losses are located in Appendix C. Separate maps were created for streams undermined in the 2nd and 3rd Act 54 assessment periods that continue to exhibit mining-induced flow loss impacts. Because these streams continued to receive augmentation during the 4th assessment, they were identified as still exhibiting flow loss.

Unfortunately, the University was unable to complete Task 3 because the paper files that were made available at the CDMO did not contain maps for the vast majority of pooling impacts. Technical drawings of pooling were available in the plans for gate cut mitigation; however, these technical drawings cannot be spatially geo-referenced to identify the pooling location. Furthermore, because they are focused on the gate cut restoration area, they often do not show the entire length of pooling, making it impossible to quantify the maximum/minimum pooling lengths. According to PADEP, maps of pooling are not required because pooling is predicted up front in the permit application. While the location and lengths of pooling could not be determined, the location and lengths of gate cut restoration areas are described below (Section VII.I.2). Gate cuts are generally necessitated by pooling and thus give some indication of pooling frequency and location.

VII.F – PADEP System for Tracking Stream Impacts

The above data and previous Act 54 reports (Conte and Moses 2005, Iannacchione et al. 2011) demonstrate that longwall mining can cause flow loss and pooling impacts to undermined streams. PADEP is responsible for "documenting observations regarding mining-induced changes" and "determining when a stream has recovered" (PADEP 2005a). Here, the system for tracking stream impacts during the 3rd Act 54 assessment period is described along with the changes that were made to that system during the 4th assessment.

During the 3rd Act 54 assessment period and prior to the adoption of TGD 563-2000-655, a "stream investigation" was initiated each time a stream impact was reported by a mining company, a property owner, or a PADEP surface subsidence agent (SSA). Each stream investigation was given a unique identifier (e.g. ST0501 represents the first (01) stream (ST) impact that occurred in 2005 (05)) that could be tracked in both BUMIS and the paper files at CDMO. Once a stream investigation was filed, PADEP would determine if the impacts were mining-related or the result of seasonal variations in climate. If the changes were determined to be mining-induced, then the mine operator was granted a period of time to perform mitigation. The operator would later submit data to PADEP for determination of stream recovery. If the stream had recovered, PADEP would release the stream from further monitoring. If the stream had not recovered, PADEP would require additional mitigation. This stream tracking procedure is summarized in Figure VII-6a.

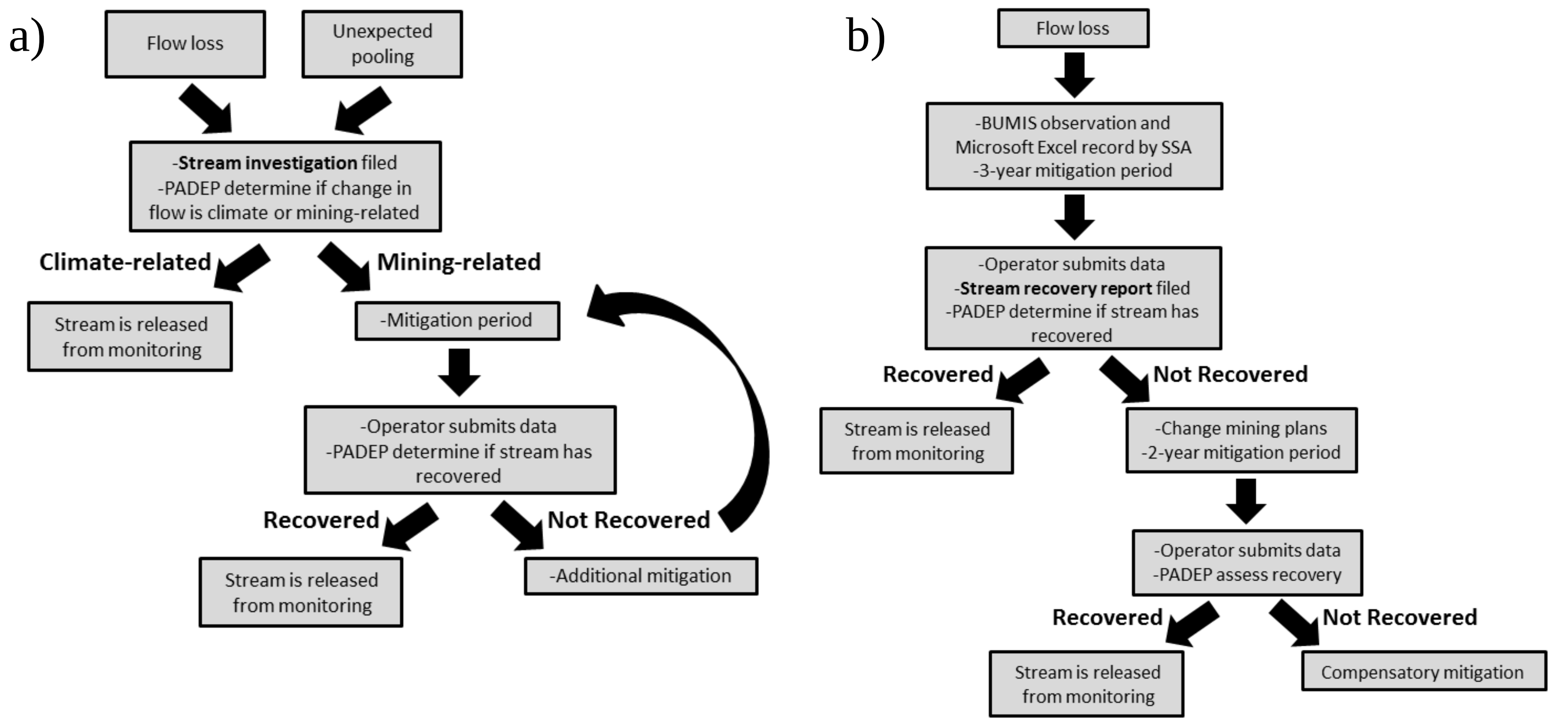

Figure VII-6. PADEP methodology for tracking stream impacts during the a) 3rd and b) 4th Act 54 assessment periods.

According to PADEP, stream investigations are still used to track impacts that occurred prior to mine operators coming into compliance with TGD 563-2000-655. However, the procedure for tracking stream impacts after this time point is quite different (Figure VII-6b). PADEP no longer has a stream investigation period in which they determine if changes in flow are related to mining or climate. Instead, changes in flow that occur at the time of mining are automatically assumed to be mining-related. If a change in stream flow is observed by a PADEP SSA, a record of the impact is made in the BUMIS agent observation files and in the SSA's stream data logs. Stream data logs are Microsoft Excel files that are stored on PADEP servers. The SSA then notifies the mine operator of the impact, and the operator has three years from the date of impact to repair the reported stream damages. Over this period, the SSA monitors the stream and tracks any mitigation work that is performed. Records of mitigation are stored in the stream data logs. If PADEP finds that stream flow and biology have not recovered after three years, then they require a change in future mining plans to avoid mining under similar settings. At this point, the operator has two more years to perform additional mitigation work. Once the operator submits the final flow and biological data to PADEP, a "stream recovery report" is generated. Stream recovery reports are paper files at CDMO that contain the data from the mine operator and reviews by PADEP hydrologists and biologists. If the flow and biology match pre-mining conditions or those of an approved control stream, the stream recovery report is closed. However, if a stream cannot be restored after a total of five years, then the operator may be required to perform compensatory mitigation. (PADEP 2005a). Compensatory mitigation generally refers to "restoration or enhancement of an equivalent length of another stream in the same watershed or a nearby watershed." (PADEP 2005a). It should be noted that the timeframe outlined here applies to flow loss impacts only – the three year recovery period outlined by TGD 563-2000-655 does not apply to unexpected pooling impacts.

Comparison of figures VII-6a and b reveals significant differences in PADEP's approach to tracking stream impacts between the 3rd and 4th Act 54 assessment periods. The 4th assessment protocol makes two significant improvements over the 3rd assessment protocol – operators are automatically assumed liable for changes in stream flow that occur at the time of mining and streams are no longer mitigated for an indefinite period of time. In terms of mitigation, TGD 563-2000-655 makes it clear that "if a stream cannot be fully restored within five years", then compensatory mitigation may be required (PADEP 2005a). While the University recognizes the advances made by PADEP in tracking stream impacts, there are several drawbacks to the 4th assessment protocols. First, tracking the occurrence and resolution of stream impacts is no longer a simple matter of tallying the number of stream investigations and their final resolution statuses. Currently, impacts are tracked by two different systems – BUMIS observation files and SSA stream data logs. The BUMIS observation files are standardized across agents and can be quite informative. However, they are written as narratives that makes extraction of relevant data difficult and time-consuming. PADEP acknowledge that BUMIS was never intended for tracking stream impacts. As a result, the SSA stream data logs were created. These logs are also a wealth of information, but there is no standardized format for the logs across the agents. Furthermore, because there is no record of these files in the filing system at CDMO, it would be difficult for citizens of the Commonwealth to request and/or obtain these data. The University was not aware that these files existed until one year after the start of the contract period. Once it was discovered that PADEP was internally tracking stream impacts on Microsoft Excel spreadsheets, the University had to specifically request them from PADEP. The University suggests that PADEP develop a centralized and standardized information system for tracking stream impacts. Maps, photos, narratives, and raw data that the SSAs are already collecting could be integrated into a system under a unique identifier for each stream. Such a system would increase PADEP efficiency and transparency to the public. The second drawback to the 4th assessment protocol is that after mitigation, PADEP relies on the mine operator to submit flow and biology data. The University suggests that PADEP specifically request these data after mitigation to ensure a timely assessment of recovery. The University further suggests that all flow and biology data collected by the mine operators during the mitigation period be required for submission to avoid any perception of selective data submission. The University is not suggesting that mine operators are engaging in such practices, only that the current practice leaves both the operators and PADEP susceptible to such suspicions. Lastly, while TGD 563-2000-655 allows for three years of restoration before requiring a change in the mining plans, it is possible that similar streams may be undermined during this time period and impacted in a similar manner. While the University has no suggestions for avoiding this inherent lag time in protecting similar streams from the impacts of mining, this situation does highlight the importance of understanding factors that contribute to stream impacts so that operators can better predict them before they even occur.

It is important to note that the PADEP approaches to tracking and record-keeping outlined above have no formal documentation associated with them (G. Shuler, PADEP, email comm.) They are not therefore policy, but only practice promulgated by word of mouth. The University recommends that a written policy be developed.

VII.G – Stream investigation and recovery reports initiated during the 4th assessment

The stream investigation and recovery reports initiated during the 4th Act 54 assessment period are described below. The number of cases, their time to resolution, and final resolution status are compared to data from previous assessments. Lastly, several case studies are highlighted to detail the current methods utilized by PADEP to assess stream recovery.

VII.G.1 – Stream investigation and recovery report data collection

Stream investigations were tracked using BUMIS and paper files at CDMO. The University collected information on the date the impact occurred, the final resolution status, and the final resolution date. Using these data, the time to resolution in days was calculated for each stream investigation.

During the course of data collection, the University discovered that BUMIS is incomplete. Two of the nine stream investigations from this period were not tracked in BUMIS – only in paper files at CDMO. For data on stream recovery reports, the University had to rely exclusively on the paper files at CDMO which are assumed to be complete.

VII.G.2 – Stream Investigations: Resolution status and time to resolution

A total of nine stream investigations were filed during the 4th assessment period (Appendix G1). Of these, five had a final resolution as of 20 August 2013 (Table VII-8). Considering that the 3rd Act 54 assessment period had a total of 55 stream investigations (Iannacchione et al. 2011), stream investigations at PADEP have declined by 84%. The decline is largely a result of the shift in methodology for tracking stream impacts (Figure VII-6). Stream impacts are now tracked in BUMIS and SSA stream data logs.

Table VII-8. Resolution status of stream investigations initiated in the 4th assessment period.

Resolution Status	Number
Final: Not recovered: compensatory mitigation required	1
Final: No actual problem	1
Final: Not due to underground mining	2
Final: Stream Recovered	1
Interim: Not Yet Resolved	4
Total	9

The small number of resolved cases from this assessment period, coupled with the wide variation in time to resolution, resulted in a large standard deviation around the mean time to resolution (776 days +/- 1050 days). Thus, any comparison with the average time to resolution from the 3rd Act 54 assessment is difficult to interpret and relatively meaningless. However, it should be noted that three of the five stream investigations were resolved within one year of the stream impact (Appendix G1).

For these five resolved stream investigations, PADEP first determined if the changes in stream flow were climate or mining-related (Figure VII-6a). For two cases during the 4th assessment, PADEP attributed flow losses to climatic conditions (i.e. drought, seasonal intermittent flow). These two cases have a final resolution of "Not due to underground mining". Close observation however revealed that inadequate data and observations served as the basis for these determinations. For case ST0902, PADEP relied on just six months of pre-mining flow data to establish the normal range of conditions for an unnamed tributary to Whiteley Creek (stream 41257) rather than using a control stream. It should be noted that this stream was undermined between 20 April and 4 May 2009 and experienced flow loss well after the mine operator was in compliance with TGD 563-2000-655. It is unclear why this case was not handled using the newer methodology developed by PADEP (Figure VII-6b) and held to the strict recovery standards set forth by TGD 563-2000-655. For case ST0903, flow loss was observed on Cessna Run (stream 46501) in TJS 6, a room-and-pillar mine. While stream flow losses are rare in room-and-pillar mines, pillar failure can occur and cause fracturing of the aboveground rock strata. The initial flow loss was reported by a property owner on 11 September 2009. A site visit was made a year later in September 2010 by PADEP. At that time, Pennsylvania was experiencing a severe drought (Table VII-2). PADEP noted that the stream was dry at this time, but that subsequent stream visits showed flow in the stream channel. Because there is no evidence that the site was visited by PADEP prior to the drought event, the cause of the 2009 flow loss – a year in which the state did not experience drought conditions – is unknown. PADEP did not request that the mine operator select a control stream and post-mining flow data was not available in the stream investigation file. Because PADEP observations indicated that flow returned following the 2010 drought, the case was closed on 9 June 2011.

PADEP also determined for one stream investigation during the 4th assessment that there was “no actual problem” (Table VII-8). In this case (ST1102), comparisons of flow data from a control stream and stream 32736, an unnamed tributary to Templeton Fork, revealed no difference in the average percentage of stream dry. Interestingly, in these analyses, only flow data from the uppermost reaches of the control stream were utilized because these reaches were most similar in drainage area to Stream 32736.

Only two cases in the 4th assessment were ruled to be related to mining. For case ST1201, PADEP determined that stream 41250, an unnamed tributary to Dutch Run, had recovered to a normal range of conditions based on flow data from the mine operator. However, closer inspection of the data again indicates that the PADEP’s decision was based on inadequate flow data. A control stream was not used to establish the normal range of flow conditions. A field visit was conducted on 3 February 2012 during which the stream was flowing and according to PADEP agents, “appears to be restored to support or sustain its uses.” For the other case (ST1203), the PADEP ruled that stream 32596, an unnamed tributary to North Fork of Dunkard Fork, is not recovered and that compensatory mitigation is required. This stream investigation combined three previous claims for stream 32596, including ST0428, ST0502, and ST0512. See Section VIII for more information on PADEP’s decision on this and other streams in Bailey Mine.

As for the four stream investigations that remain open, pre-mining flow data or control stream flow data have been submitted for three of the cases (ST0901, ST1001, and ST1101). PADEP hydrologists are currently reviewing these data to make a final decision regarding the stream status. However, for two of the three cases the flow data that is currently available is inadequate. For case ST1001, only eight months of pre-mining flow data is available. A control stream has not yet been proposed to more thoroughly establish baseline conditions. For case ST1101, flow data from the control stream were collected between September 2008 and May 2010. The mine operator compared these data to post-mining data on the impacted stream that was collected between September 2007 and December 2008. The time periods for comparison do not align, thus negating the usefulness of the control stream comparison. The final unresolved case, ST1202, involves stream 32719, an unnamed tributary to Rocky Run. According to the stream investigation file, the PADEP required the mine operator to perform mitigation work to restore flow conditions to the stream. The mine operator submitted restoration plans in July 2010 to perform shallow grouting and remove any alluvial bedload material that was impeding stream flow. Unfortunately, there was no additional information available on this stream investigation in the paper files at CDMO. As a result, the University could not determine if the mitigation work had ever been completed and if so, if it was successful.

VII.G.3 – Stream Recovery Reports: Resolution status and time to resolution

A total of 14 stream recovery reports were submitted by mine operators to PADEP during the 4th assessment period (Table VII-9; Appendix G2). PADEP has reached a final resolution on 11 of these cases, with only three reports remaining in review by PADEP agents (Table VII-9). Of the resolved cases, nine were released from monitoring. However, two cases from Bailey Mine require compensatory mitigation (stream 32511, an unnamed tributary to Dunkard Fork, and Crow's Nest, an unnamed tributary to North Fork of Dunkard Fork; see Section VIII for more information on the PADEP's decision on this and other streams in Bailey Mine).

Table VII-9. Resolution status of stream recovery reports from the 4th assessment period.

Resolution Status	Number
Final: Not recovered: compensatory mitigation required	2
Final: Released	9
Interim: Not Yet Resolved	3
Total	14

Inspection of the flow and biological data that were submitted by the mine operator and used by the PADEP to assess stream status revealed that data were largely collected in accordance with TGD 563-2000-655. The University noted only two problems with the data submissions. First, in several cases, just one post-mining TBS was submitted and compared to control stream biology to determine recovery (e.g. SR0901, SR0905, SR1001). Often the lone post-mining TBS was collected by PADEP biologists. TGD 563-2000-655 clearly states that recovery must be based on a *mean* post-mining TBS that is generated from at least two TBS samples that are within 16% of each other (PADEP 2005a). Second, pre-mining flow measurements in one case were not made on a weekly basis for 6 months prior to mining. A control stream should have been used for comparison with post-mining flow conditions, but there was no evidence that a control stream was used (SR1001). Overall, the data associated with the stream recovery reports appears to be much more detailed and more in line with PADEP policy than that found in the stream investigation files. However, biological recovery assessments must be based on two post-mining samples – not just one. The University also suggests that the mine operators, rather than PADEP agents, should be responsible for generating these post-mining scores.

VII.H – Biological Assessment of Streams Impacted by Flow Loss and Pooling

To determine the biological impacts of flow loss and pooling impacts, PADEP tasked the University with assessing pre- and post-mining Total Biological Scores (TBS) for at least five stream segments that experienced mining-induced flow loss and at least five stream segments that experienced mining-induced pooling. The University's findings are presented below.

VII.H.1 – Stream Biology Data Collection and Analysis

Data on stream biology is stored exclusively in the paper files at CDMO. While the University found an abundance of pre-mining biology data in the paper files, post-mining biology data was generally scarce and difficult to find.

For mining-induced flow loss impacts, the University was only able to locate pre- and post-mining TBS for streams from one longwall mine – Bailey Mine. During the permitting of the Bailey Mine South Expansion, PADEP developed a compliance schedule that listed dates for the submission of pre- and post-mining stream and wetland data. These data submissions were located in a folder labelled “Bailey Mine TGD” at CDMO. To the University’s knowledge, no other longwall mine has been placed under a compliance schedule by PADEP and therefore similar data were largely unavailable from other mine operators (see below). In addition to the data located in this file, additional pre- and post-mining TBS data were located in the stream recovery files. In total, the University identified 24 stream bio-monitoring stations in Bailey Mine that had experienced mining-induced flow loss impacts (i.e. received augmentation and/or grouting) and had both pre- and post-mining TBS data. While these data are likely not representative of flow loss impacts at all longwall mines, pre- and post-mining TBS from other mines were not made available to the University.

For mining-induced pooling impacts, biology data is available in the stream restoration reports that are submitted annually to PADEP following the gate cut. However, most of the TBS data in these reports is collected either pre-mining or post-restoration. It seems that in general, very few TBS are collected after mining, but before the gate cut occurs. TGD 563-2000-655 stipulates that mitigation plans for pooling should be designed to address mining induced changes before they result in adverse effects on streams and wetlands (PADEP 2005a). The short time to gate cut mitigation in several mines (Figure VII-13) suggests that gate cuts are being performed rapidly to prevent such adverse effects from occurring. As a result, pre- and post-mining (pre-mitigation) TBS data were limited to eight stream bio-monitoring stations in two longwall mines – Bailey and Enlow Fork Mine.

Prior to analysis, datasets were corrected for spatial and temporal correlations among the observations. An important assumption of statistical analysis is that observations are uncorrelated and independent of one another. To correct for spatial correlation among the bio-monitoring stations, the University eliminated all stations that were within 1,673-ft of another station (see Section VII.C.2 for methodology). This correction removed three stations from the flow loss dataset and four stations from the pooling dataset. While it might seem that repeated measures analysis should be used to correct for temporal correlations since the data are longitudinal in nature (i.e. a single bio-monitoring station is sampled at multiple time points and samples that are collected closer in time may be more similar than samples collected over distant time points), samples were not taken at equally spaced time points across sampling stations. The equal spacing of time points across sampling stations is a requirement of repeated measures. Thus a repeated measures approach was precluded. This analysis does not distinguish among the different pre- and post-mining time points – instead it tests how, on average, post-mining TBS differ from pre-mining TBS. The University analyzed raw pre- and post-mining TBS and as well as adjusted TBS that were corrected for the influence of catchment and reach-scale characteristics as well as month of sampling (see Sections VII.C.2 and VII.C.3). For mining-induced flow loss impacts, the final dataset was large enough ($N = 21$ stations with 125 TBS samples) to use analysis of variance (ANOVA) to statistically test the difference between pre- and post-mining scores (model: raw or adjusted TBS = mining (pre-mining vs. post-mining) + station). For mining-induced pooling impacts, the remaining dataset was not large enough to meet the assumptions of a statistical analysis ($N = 4$ stations with 20 TBS samples). Instead, the average pre- and post-mining scores and their standard errors are reported.

Regarding the lack of post-mining biology data, PADEP asserts that data was requested from Cumberland and Emerald Mines in April 2013. Data were submitted by the mine operator on 7

August 2013. However, PADEP did not start review of the data at this time or notify the University of its existence because the data were misplaced in CDMO following submission (G. Prentice, PADEP, email comm.). Once the data had been found, PADEP begin its review in February 2014. At this time, the University was still not made aware of the data and did not discover it until April 2014. Upon reviewing the data in April, it became evident that the Cumberland and Emerald stream biology data lacks compliance with TGD 563-2000-655. First, very few stations had pre-mining TBS. Of the 52 bio-monitoring stations in the report, only 11 had at least two pre-mining TBS, a requirement of the TGD. While stations undermined shortly after the TGD's implementation may be expected to lack sufficient data, stations undermined as recently as 2010 in both Cumberland and Emerald Mines reported just a single pre-mining TBS. As a result, the bulk of the post-mining biology data for Cumberland and Emerald Mines are compared to data from control streams that were chosen post-mining. Second, the pre-mining data that is available suggests that the consultants employed by Alpha Natural Resources, Inc. are not following the sampling protocols required by Appendix B of TGD 563-2000-655. The pre-mining scores are highly variable – of the 11 sites with two pre-mining TBS, just five sites had scores that met the TGD 563-2000-655 requirement of being within 16% of each other. PADEP discovered that consultants were not using 200 organisms +/- 20% to generate TBS. PADEP agents have trained consultants on TGD 563-2000-655 protocols both in the field and in the lab (PADEP, pers. comm.). Lastly, the month of sampling (as well as data on water quality and macroinvertebrate community composition) was only reported for samples collected on control streams. Without this information, scores cannot be corrected for temporal variation in sampling date. Due to uncertainty regarding the validity of the data for the reasons outlined above, the University did not include any stations from Cumberland or Emerald Mine in the assessment of mining-induced flow loss and pooling impacts on stream biology. The problems encountered suggest that the PADEP would be well served to require certification of consultants' abilities at sampling according to protocols and competency at macroinvertebrate identification. Furthermore, the University recommends that PADEP closely evaluate the data submissions that are made available before and after mining.

PADEP also acknowledges that post-mining TBS were requested from Enlow Fork Mine in February 2014. Data were submitted by the mine operator in March 2014 (G. Prentice, PADEP, email comm.), however these data were never made available to the University.

VII.H.2 – Pre- and Post-Mining Total Biological Scores from Streams Impacted by Mining-Induced Flow Loss and Pooling

On average, mining-induced flow loss reduces a stream's Total Biological Score by 9 points (Figure VII-7). Despite a small sample size, this result is highly significant (effect of mining on raw TBS, $F_{1,103} = 16.44$; $P < 0.0001$; effect of mining on adjusted TBS, $F_{1,103} = 15.84$; $P = 0.0001$). The reduction in TBS is greater than 12% of the pre-mining average, indicating that, on average, mining-induced flow loss has an adverse effect on stream biological communities (PADEP 2005a).

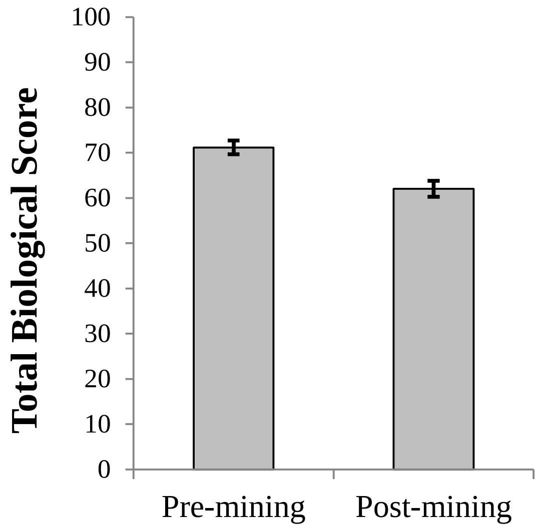

Figure VII-7. Comparison of pre- and post-mining Total Biological Scores for streams impacted by mining-induced flow loss. Data are least squares means +/- 1 standard error.

A similar pattern was found for streams impacted by mining-induced pooling. On average, pooling reduces a stream's adjusted TBS by 7 points (pre-mining adjusted TBS: 67.7 +/- 2.9 vs. post-mining adjusted TBS: 60.5 +/- 6.2; means +/- 1 standard error). When looking at the unadjusted scores, pooling reduces raw TBS by 5 points (pre-mining raw TBS: 63.5 +/- 2.3 vs. post-mining raw TBS: 58.4 +/- 5.9). As with flow loss, the reduction in adjusted TBS is greater than 12% of the pre-mining average. However, the reduction in raw TBS is within 12% of the pre-mining average. While these results suggest that adverse effects are occurring on pooled segments before gate cut mitigation work begins, additional data would allow a more definitive conclusion.

Overall, these two analyses demonstrate that mining-induced flow loss and to a lesser degree, mining-induced pooling, have significant detrimental effects on stream communities. While the data came from a limited number of longwall mines, the results are concordant with reported reductions in macroinvertebrate taxa richness following longwall mining reported elsewhere (Stout 2003).

VII.H.3 – Changes in Macroinvertebrate Community Composition for Streams Impacted by Mining-Induced Flow Loss

Total Biological Score is a multi-metric index of stream community health (PADEP 2005a). While it is useful in measuring overall changes in community status, it does not explain how the community changes functionally or taxonomically in response to disturbance. To understand how mining affects macroinvertebrate communities in terms of taxa and function, the University investigated changes in community composition. Unfortunately, data on community composition are not always reported to PADEP. Often, mine operators submit just the TBS and its five associated metrics to describe pre- or post-mining stream samples. Within the flow loss dataset, only 16 stations had samples that reported pre- and post-mining macroinvertebrate sample composition (N = 28 pre-mining samples, N = 34 post-mining samples). For these samples, the

University calculated the relative frequency of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa occurrences. The University focused on these macroinvertebrate orders because EPT taxa are often thought to be more sensitive to disturbance (although this is not always true; Rosenberg et al. 2008) and EPT richness or % EPT is a commonly used metric in many multi-metric stream indices (e.g. Gerritsen et al. 2000, Burton and Gerritsen 2003), including the Total Biological Score (PADEP 2005a). Non-metric multi-dimensional scaling (NMDS) was also used to visualize community similarity across pre- and post-mining samples (R, version 3.0.2; R Core Team 2014). NMDS is a type of ordination analysis. Ordination reduces multivariate data, such as species abundances, to a smaller number of composite variables. Each sample receives a score for these new composite variables and the samples can then be plotted in multi-dimensional space. The scores preserve the degree of differentiation between the samples so that samples that are far apart in graphical space differ from each other, while samples that cluster together in graphical space are similar to each other. Data were log-transformed ($\log_{10}(x) + 1$) prior to ordination. Following the ordination, permutation tests were used to determine if time of sampling (pre- vs. post-mining) was a significant predictor of community composition.

The majority of Ephemeroptera taxa declined in frequency following mining-induced flow loss (Figure VII-8). Many taxa experienced dramatic declines, with eight genera showing a ~50% reduction in their occurrence (Figure VII-8). The most striking reductions are found in *Ephemerella*, *Eurylophella*, and *Epeorus*. These taxa were relatively common prior to mining, occurring in 50-60% of all samples. After the flow loss disturbance created by mine subsidence, these taxa became quite rare and were found in 26% or less of the samples. In general, the families Ephemerellidae and Heptageniidae appear to be highly sensitive to flow loss, with all genera in these families experiencing declines following mining (Figure VII-8; exception: *Nixe* in Heptageniidae).

In contrast, Plecoptera and Trichoptera appear to be quite robust to mining-induced flow loss, as few taxa experienced strong declines in frequency following mining (Appendix H1). Many taxa remained at a similar frequency or even increased in frequency. For example, the Trichoptera genus *Ironoquia* occurred at just 5% of sites prior to mining but was present at nearly 50% of sites after mining-induced flow loss occurred (Appendix H1). Similarly, the genus *Isoperla* doubled in frequency following mining-induced flow loss (Appendix H1). In general, taxa from these orders appear to be more tolerant of mining-induced flow loss.

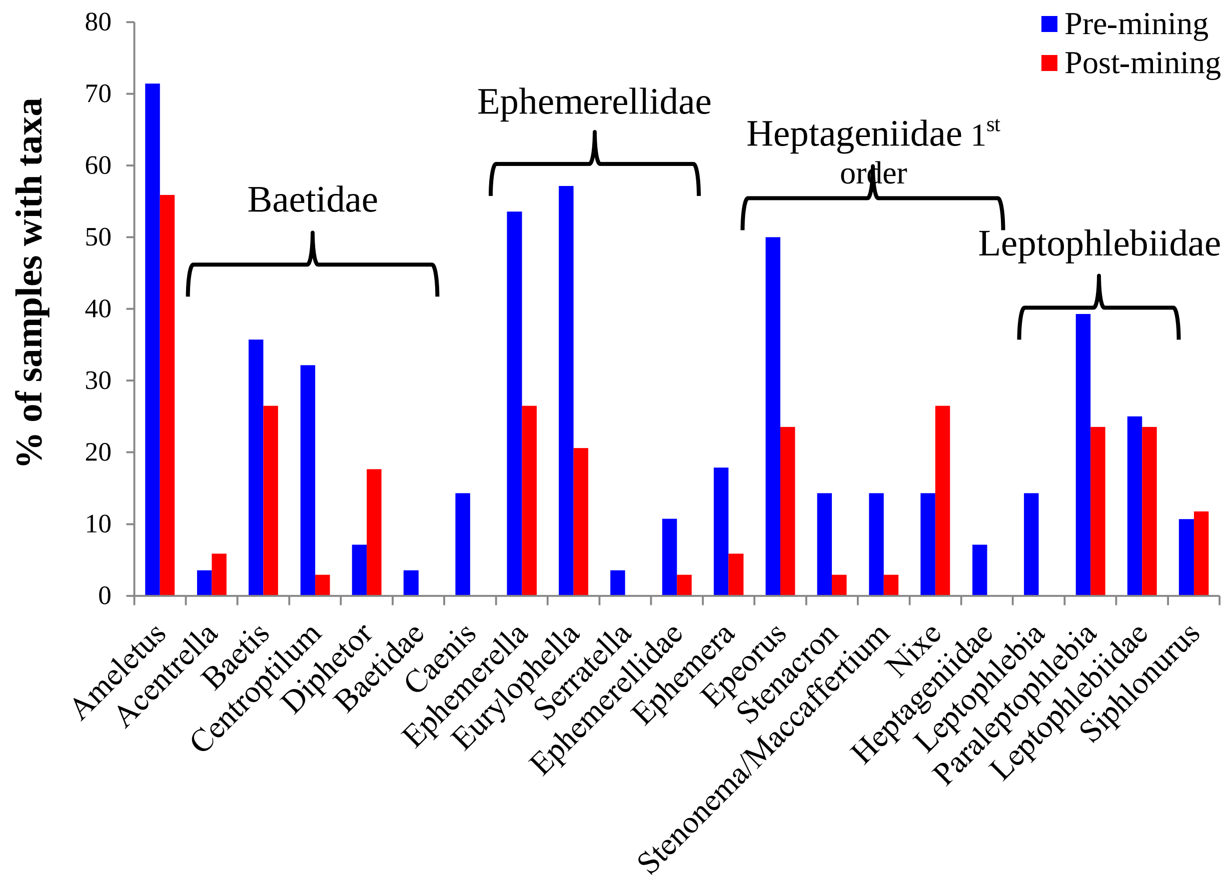

Figure VII-8. Relative frequency (%) of Ephemeroptera taxa occurrences in pre- (N = 28) and post-mining (N =34) samples. While the bulk of the samples were identified to the genus level, consultants were at times only able to identify to the family level. For families with more than one genus represented, a bracket is used to group all genera belonging to that family.

NMDS shows that community composition differs between pre- and post-mining samples (ordination stress = 0.23; Figure VII-9). Permutation tests confirm this difference, as time of sampling was a highly significant predictor of community composition (R^2 = 0.15; P = 0.001). Month of sampling was also a significant predictor of community composition (R^2 = 0.62; P = 0.001), but station id was not. The post-mining communities are shifted towards the top of Figure VII-9 compared to the pre-mining communities. Functionally, this reflects a shift away from scraper and collector-gatherer taxa, such as the Ephemeroptera discussed above, towards the shredder and predator taxa of the Plecoptera and Trichoptera. Stress tolerant Diptera taxa, including *Ormosia*, *Simulium*, and *Clinocera*, also appear to be more highly correlated with post-mining communities than pre-mining communities.

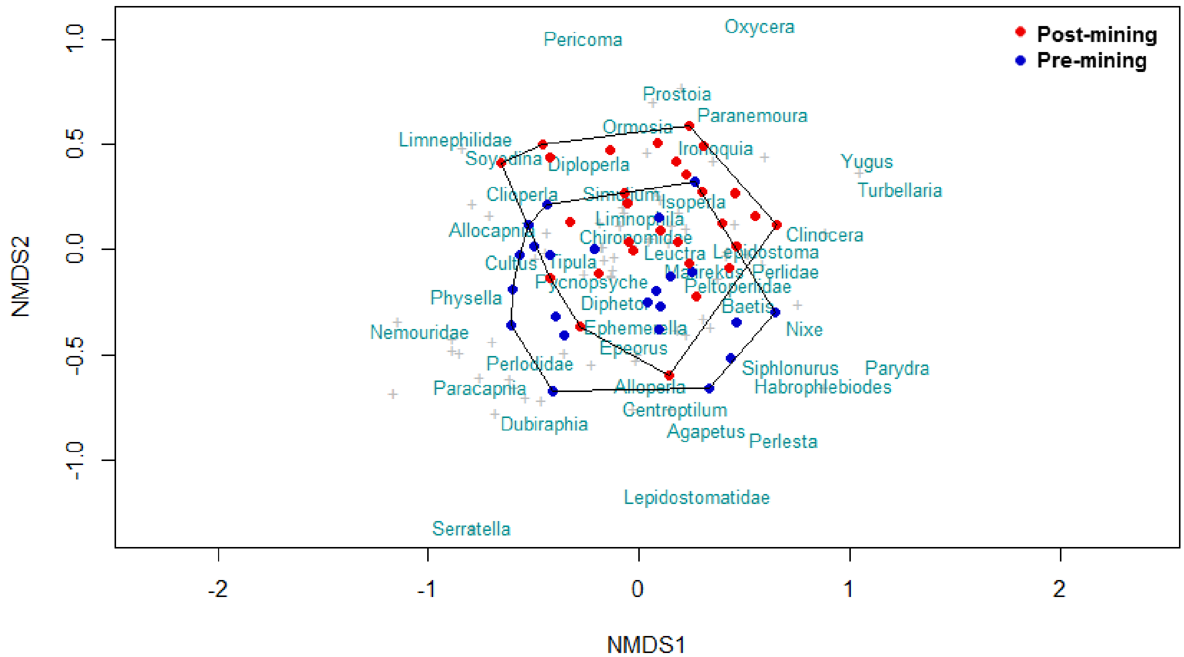

Figure VII-9. Non-metric multi-dimensional scaling ordination for community composition of pre- (N = 28) and post-mining (N = 34) samples. Gray plus signs represent additional taxa whose names are not displayed. When taxa overlap, only the name of the more abundant taxa is displayed.

These results are similar, although less dramatic, to the effects of mountaintop coal mining on stream communities (Pond et al. 2008). In mountaintop coal mining, layers of overburden are placed in valleys near the surface mine. The valley fills not only affect the physical habitat of intermittent and perennial streams in the valleys, but they can also affect the quality of groundwater and surface runoff entering the streams. Mountaintop coal mining is not currently practiced in Pennsylvania, but is used in other parts of Appalachia (Kentucky, Tennessee, Virginia, and West Virginia).

Pond et al. (2008) found that stream communities of mined sites in West Virginia were distinctly different from those of unmined sites. In particular, the community level differences were strongly linked to the loss of taxa from the families Ephemerellidae and Heptageniidae in mined sites (Pond et al. 2008). While the physical disturbance of mountaintop mining (i.e. stream burial) is dramatically different from the disturbance associated with longwall mining (i.e. streambed fracturing), it appears that the two mining methods can have similar impacts on aquatic communities. However, the complete loss of many Ephemeroptera taxa from mountaintop mined sites in the Pond et al. (2008) study indicates that the effects of mountaintop mining on stream biology are much stronger than those of longwall mining.

The loss of Ephemeroptera at mountaintop mining sites has been linked to increases in specific conductivity (hereafter, conductivity; Pond et al. 2008, Pond 2010). Conductivity is defined as the ability of a material to conduct an electrical current (µS/cm) at a standardized temperature (25°C). For streams, conductivity reflects the concentration of charged particles in the water. Because conductivity has been associated with declines in macroinvertebrate health in the Western Allegheny Plateau ecoregion, the U.S. EPA recently established a conductivity benchmark for aquatic life in this area (U.S. EPA 2011). A benchmark of 300 μS/cm was selected because analyses indicate that 95% of aquatic taxa can persist at this level. While the benchmark was not derived using data from Pennsylvania, it is expected to be applicable to sites in Greene and Washington counties because they are part of the Western Allegheny Plateau ecoregion. To determine if the changes in Ephemeroptera frequency following longwall mining (Figure VII-9) are similarly related to changes in conductivity, the University analyzed pre- and post-mining chemistry data. Within the flow loss dataset, 17 stations had samples that reported pre- and post-mining conductivity, pH, dissolved oxygen, and stream flow. The data were collected concurrently with biological sampling. Using these parameters, flux was also calculated. Because surface runoff can dilute the concentration of charged particles in the stream, flux adjusts the conductivity measures by flow rate (flux = conductivity*flow). All metrics were first analyzed using a multivariate analysis of variance (MANOVA; model: conductivity pH dissolved oxygen flow flux = mining station) to assess the overall effect of mining on stream physiochemistry. Following a significant MANOVA, individual two-way ANOVAs were used to test the effect of mining and station identity on each metric. Metrics were log-transformed when necessary to meet assumptions of normality and homoscedasticity. One post-mining conductivity measure was dropped from analyses because studentized residuals indicated that it was a significant outlier (947 μS/cm).

Stream physiochemistry is significantly affected by mining (MANOVA, Roy's greatest root = 1.50, F = 28.82, numerator degrees of freedom = 4, denominator degrees of freedom = 76, $P < 0.0001$). This result was driven in large part by significant increases in conductivity ($F_{1,80}$ = 106.21, $P < 0.0001$) and pH ($F_{1,81}$ = 67.60, $P < 0.0001$) following mining (Table VII-10). Unfortunately, data for flow and flux could not be analyzed using individual ANOVAs because despite transformation, the residuals violated test assumptions. However, the mean post-mining flux is ~3 times larger than the pre-mining value (Table VII-10), indicating that even after adjusting for large flows during storm events, post-mining streams had elevated levels of dissolved materials in the water. In contrast, the level of dissolved oxygen did not differ between pre- and post-mining samples (Table VII-10).

*Table VII-10. Pre- and post-mining physiochemical parameters for streams experiencing flow loss. Following a significant MANOVA, each parameter was analyzed using a two-way ANOVA to test the effect of mining and station identity. * = significant effect of mining, P < 0.05. N = sample size, DO = dissolved oxygen. Flux = conductivity*flow.*

	Pre-mining				**Post-mining**			
	Mean	Std. Error	Range	N	Mean	Std. Error	Range	N
Conductivity*	169.1	9.1	58-416	61	329.9	12.3	148-462	37
pH*	7.30	0.04	6.55-8.1	61	7.88	0.07	6.7-8.69	38
DO	10.9	0.3	7.1-15.9	61	10.7	0.3	5.62-13.84	38
Flow	0.18	0.03	0.001-1.1	61	0.28	0.07	0.01-2.99	45
Flux	26.9	4.7	0.11-180	61	80.2	19.5	1.57-442	37

Elevated conductivity and alkalinity levels have previously been observed in streams over longwall mining (Stout 2003). These changes were attributed to increased retention time underground (Stout 2003). Extra time spent in underground rock layers could increase contact time between the water and the rock and allow more dissolved solids to enter the water.

However, at sites experiencing flow loss, grout mitigation (see Section VII.I.3) and subsequent weathering of the grout material could also lead to increases in conductivity and pH. Unfortunately, the available data cannot be used to determine the exact mechanisms underlying the increases in conductivity and pH observed here. It is also unclear if the increased conductivity is directly responsible for declines in Ephemeroptera frequency at longwall mined sites. Indeed, the nature of the relationship between conductivity and biotic integrity remains the subject of much investigation (U.S. EPA 2011). However, longwall mining clearly pushes stream conductivity levels over the U.S. EPA benchmark for aquatic life (Table VII-10). The University suggests that PADEP and future Act 54 reports further investigate the impacts of longwall mining on stream water quality in the Commonwealth.

Changes in community composition could not be assessed for streams with mining-induced pooling impacts because the available data only had two stations that reported pre- and post-mining macroinvertebrate taxa abundance. This sample size is not sufficient to draw conclusions regarding the impact of subsidence related pooling on stream macroinvertebrate community structure.

VII.I – Stream Mitigation

Following subsidence-induced impacts, mine operators employ a variety of techniques to repair streams. For pooling impacts, TGD 563-2000-655 requires that mitigation be performed when the pool depth increases exceed 1-foot or more, or when other adverse conditions are created (e.g. loss of riffle habitat, sedimentation, nuisance to property owners; PADEP 2005a). For predicted flow loss impacts, the mine operator must have mitigation measures in place to restore flow to the stream within 24 hours (PADEP 2005a). If the flow loss was unpredicted, then the mine operator has 15 days to restore flow (PADEP 2005a). The mitigation plan for flow loss impacts should also include measures that are designed to restore natural stream flow "within one year or within a specified time period" (PADEP 2005a). Below, the techniques that are commonly used by mine operators to mitigate stream impacts are described. The number of streams receiving mitigation during the 4th assessment is also reported along with the time to mitigation.

VII.I.1 – Gate cut and stream channel restoration methods

To mitigate pooling impacts, gate cuts lower the stream bed elevation and establish a new stream gradient to promote flow across the gate area (Figure VII-10). Gate cutting is a multi-step process that involves design, permitting, construction, and monitoring. Because models are used to predict where pooling will occur, gate cut designs are typically drawn up and submitted to the PADEP along with the mine permit application before mining even begins. Following mining, the mine operator must meet with a DEP biologist and mining engineer to investigate any pooling impacts and determine if the existing mitigation plans are adequate (PADEP 2005a). If the existing mitigation plans require modification or if the pooling occurred in unpredicted areas, a permit revision must be filed with the DEP. In addition to the mine permit, a Chapter 105 permit is also required for gate cut mitigation. Because gate cuts alter the course, current, and/or cross section of a stream, the mitigation constitutes encroachment on a body of water, a process that is regulated by PA Code, Title 25, Chapter 105. The Chapter 105 permit must be approved by both the PADEP and the U.S. Army Corps of Engineers. Once all permits are in place, construction can begin.

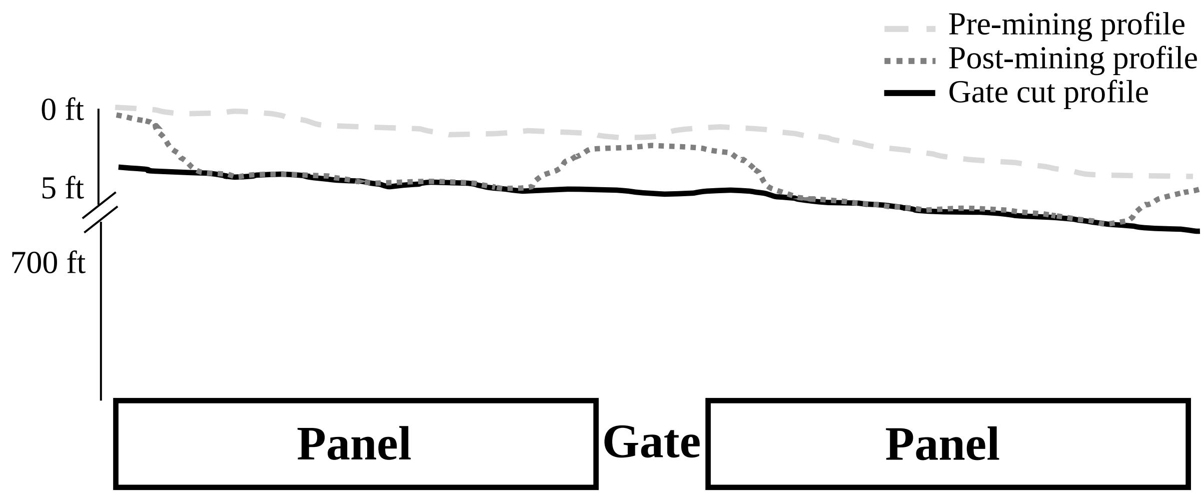

Figure VII-10. Conceptual drawing of a cross sectional stream profile before mining, after mining, and following gate cut mitigation.

To prepare the construction site, a rock construction entrance is built and silt fence is erected around equipment and staging areas to prevent sedimentation to the stream. These measures are required erosion and sedimentation controls and are outlined in Technical Guidance Document 363-2134-008 (PADEP 2012). Stream flow is routed around the restoration area using a pump and hose, commonly known as a "pump-around". To minimize the volume of water that must be re-routed, the majority of gate cuts (~85% in this assessment period) are conducted during the dry season (June-November). Fish that are observed in the stream channel are netted and moved outside of the restoration area. Once the stream bed is clear, excavation work begins at the downstream end and proceeds in an upstream direction. The alluvial material that is excavated from the stream is stockpiled nearby, and following the gate cut, some of it is placed back in the bottom of the stream to provide habitat and cover for macroinvertebrate and fish species. The remaining alluvial material is removed to a permanent disposal site.

As the channel is excavated, hydraulic control structures are installed to prevent bank erosion and promote in-stream habitat diversity. These structures include rock vanes, log vanes, and J-hook vanes, which are designed to reduce shear stress on the near bank and promote stream flow through the center of the channel (Rosgen 2001). These structures also create pools immediately downstream that provide fish habitat. Other structures, such as boulders and root wads, are also used to stabilize banks and create habitat for fish and macroinvertebrates.

Finally, the banks are re-graded to provide stability and prevent further erosion. Cut banks are typically re-graded with a minimum 2H:1V slope (i.e. 2 horizontal feet for every 1 vertical foot; PADEP 2012). Steeper slopes, which would increase flow velocity and the likelihood of erosion, are only permitted in rare cases (e.g. to preserve trees along the stream banks inside East Finley Park, Templeton Fork F10 panel gate cut; B. Dillie, PADEP, pers. comm.). Following bank regrading (Figure VII-11), live plant cuttings and previously potted vegetation are installed and the banks are seeded and mulched. The banks are further stabilized using a combination of biodegradable mats, coir logs, and other materials.

Figure VII-11. Photograph of the recently completed E20 gate cut on Crafts Creek. Banks had been re-shaped, but vegetation had not yet been planted. (Photo courtesy of A. Iannacchione).

To assess the performance and success of the gate cut, the site is monitored for a period of five years following mitigation as a condition of the Chapter 105 permit. Each year, an annual stream restoration monitoring report is submitted to PADEP. During the five year period, the mine operator must demonstrate that the post-restoration Total Biological Scores fall within 88% of the pre-mining or control stream scores (PADEP 2005a). Once the community has recovered, biological monitoring ceases. However, stream enhancement structures and vegetation are assessed on an annual basis for the full five years. The stream enhancement structures are monitored for stability, and if any have become unstable, dislodged or removed, corrective measures must be performed. For vegetation, percent cover on the banks must reach 75-85% by the end of the third year, and no planted species should comprise more than 35% of the area (CEC 2012a, CEC 2012b, WPI 2012).

VII.I.2 – Gate cut and stream channel restoration during the 4th assessment

Based on data in the permit revisions and conversations with PADEP SSA, the University determined that 28 gate cuts were performed across 4.21 miles of stream during the 4th assessment period (Table VII-11). Enlow Fork Mine had the greatest number of stream segments with gate cuts (N = 12), while Blacksville 2 Mine had the fewest (N = 0). Many streams received multiple gate cuts over this period. South Fork of Dunkard Fork in Bailey Mine and Templeton Fork in Enlow Fork Mine each had six gate cut mitigation projects totaling 7,030-ft and 4,001-ft of stream length, respectively. The single longest gate cut mitigation project occurred on Dyers Fork in Cumberland Mine (Table VII-8). Nearly 4,000-ft of stream were mitigated over the 53-54 gate road area. Prior to the gate cut, pooling along this stretch of Dyers Fork was severe with increases in natural stream depth of up to 6.1-ft along the southern edge of panel 53 (Cumberland Permit Revision 96). Several factors likely contributed to the significant amount of restoration work required on this stretch of Dyers Fork. First, Dyer's Fork has an extremely low gradient (0.015%; WPI 2012). Second, the gate road length between the 53-54 panels is nearly three times larger than typical gate roads in Cumberland Mine (686-ft vs. 216-ft), meaning that an unusually large area between the panels did not subside (Figure VII-12). Lastly, the stream runs nearly parallel to the 53-54 gate road (Figure VII-12). For all of these reasons, long stretches of stream had to be cut to reach a lower stream bed elevation and alleviate the pooling. Similarly, South Fork of Dunkard Fork in Bailey Mine runs nearly parallel to the 11I-12I gate road area and this area also required a substantial gate cut restoration project following mining (Table VII-11).

TGD 563-2000-655 indicates that mitigation of mining-induced pooling impacts should be performed before the pooling results in adverse effects on streams (PADEP 2005a). Therefore, the University predicted that gate cut mitigation would occur rapidly after mining and subsidence of the stream. The University used data from stream restoration files and SSA BUMIS observations to determine the date that each gate cut project began. The PADEP monthly flow map database was also used to roughly determine the date when each stream was undermined. A stream segment was determined to be undermined when the longwall face was completely past the stream. The difference between these two dates represented the time to restoration.

On average, it takes 682 +/- 100 (mean +/- 1 standard error) days once mining has occurred for restoration work to begin on pooling impacts. However, there is significant variation across mines (Figure VII-13). Gate cuts in Bailey and Enlow Fork Mines occur, on average, less than two years following undermining of the stream while the average gate cut in Cumberland and Emerald Mines is not initiated until nearly three years after mining (Figure VII-13).

Table VII-11. List of all stream segments receiving gate cuts during the 4th assessment period.

Mine	PA WRDS Stream Code	Stream Name	Panel	Length of Restoration (ft)
Bailey	32540	Barneys Run	12I	200
Bailey	32536	South Fork of Dunkard Fork	10I	1,100
Bailey	32536	South Fork of Dunkard Fork	11I	1,960
Bailey	32536	South Fork of Dunkard Fork	12I	1,070
Bailey	32536	South Fork of Dunkard Fork	13I	1,320
Bailey	32536	South Fork of Dunkard Fork	14I	1,180
Bailey	32536	South Fork of Dunkard Fork	15I	460
Cumberland	41261	Dyers Fork	52-53	1,007
Cumberland	41261	Dyers Fork	53-54	3,962
Cumberland	41246	Dutch Run	52	700
Cumberland	41246	Dutch Run	53	667
Cumberland	41246	Dutch Run	54	300
Cumberland	40592	Pursley Creek	58	No data
Cumberland	41178	Whiteley Creek	55-56	536
Emerald	41268	Mount Phoebe Run	B7	814
Emerald	41269	UNT to Mount Phoebe Run	B7	137
Enlow Fork	40938	Crafts Creek	E17	No data
Enlow Fork	40938	Crafts Creek	E18	600
Enlow Fork	40938	Crafts Creek	E19	837
Enlow Fork	40938	Crafts Creek	E20	977
Enlow Fork	32708	Templeton Fork	F13	950
Enlow Fork	32708	Templeton Fork	F14	600
Enlow Fork	32708	Templeton Fork	F15	600
Enlow Fork	32708	Templeton Fork	F16	700
Enlow Fork	32708	Templeton Fork	F17	626
Enlow Fork	32708	Templeton Fork	F18	525
Enlow Fork	32738	UNT to Templeton Fork	F17	375
Enlow Fork	32739	UNT to Templeton Fork	F17	50
TOTAL:				**4.21 miles**

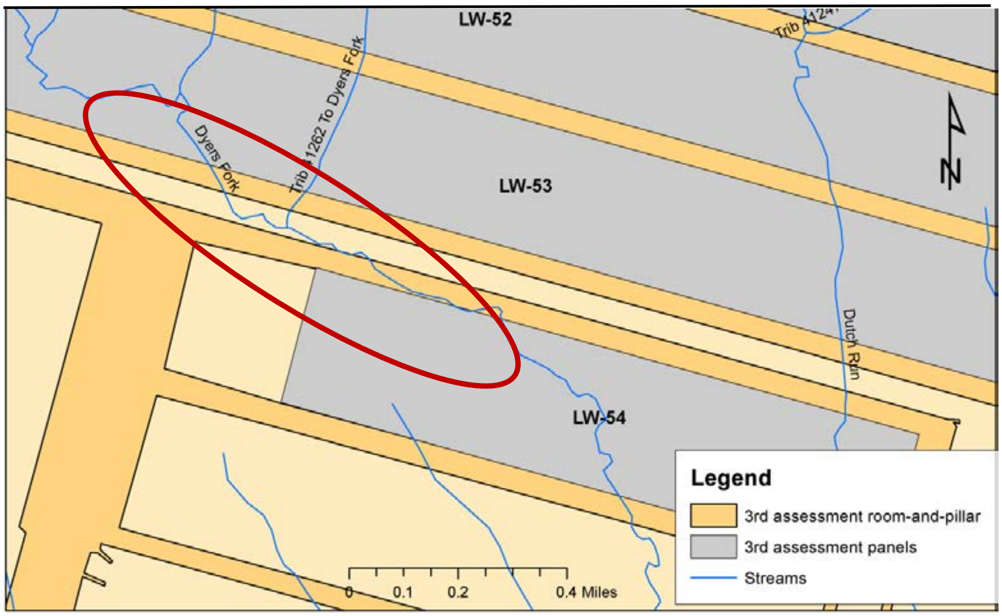

Figure VII-12. Approximate area of the Dyers Fork 53-54 gate cut is circled in red. A long gate cut was required here because the stream runs nearly parallel to the gate and the gate is wider than normal.

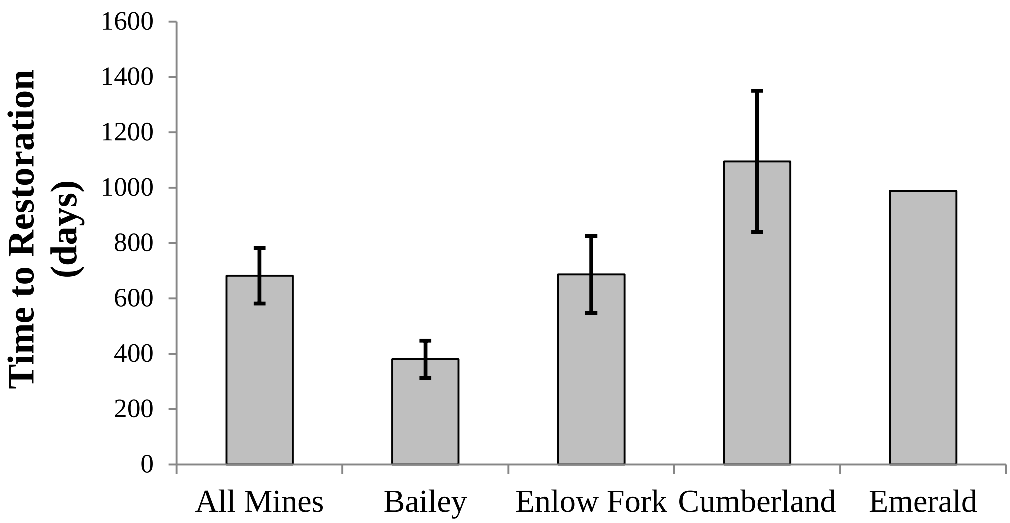

Figure VII-13. Time to gate cut and channel restoration for streams impacted by pooling during the 4th assessment. Data are means +/- 1 standard error.

Several factors may have contributed to the increased time to restoration in Cumberland Mine. First, two gate cuts involved restoration work under major bridges and thus required an additional set of approvals from PennDOT. For the Whiteley Creek 55-56 gate cut, which occurred near the I-79 bridge, confusion over the culvert type at the bridge between PennDOT and the mine operator delayed restoration for over four years after mining had occurred. For the Dyers Fork 52-53 gate cut, PennDOT expressed concern that gate cutting activities could affect the stability of the Route 19 bridge abutments and alter the hydraulic capacity of the bridge (WPI 2012). The agency had recently completed a reconstruction of the Route 19 bridge in 2006.

Ultimately, the mine operator and PennDOT reached an agreement to complete the gate cut in two phases: Phase 1 included a partial gate cut downstream of the bridge, while phase 2 finished the gate cut under the bridge and installed rip-rap along the stream bank and crossvanes in the stream channel (WPI 2012). Phase 1 was initiated roughly 336 days after mining, but phase 2 was not completed until nearly 4 ½ years after mining took place. For the analysis in Figure VII-13, the University considered the time to restoration for this gate cut to be 336 days. In addition to the complications introduced by PennDOT, two other gate cut projects in Cumberland Mine involved impacting an existing wetland. The design for gate cuts 52 and 53 over Dutch Run called for the excavation of 0.88 acres of wetland DR-27. To mitigate this wetland loss, the gate cut designs had to incorporate a wetland mitigation site. The time required to design the wetland mitigation site and obtain approvals for impacting an existing wetland may have increased the time to mitigation for these two sites as well. Overall, it appears that working with additional agencies and general design challenges may have hampered the restoration efforts in Cumberland Mine.

Gate cuts are also proposed for a number of additional sites that experienced pooling impacts during the 4th assessment period (Table VII-12). While Blacksville 2 Mine did not have any gate cuts during the 4th assessment, several are planned and awaiting approval by the PADEP. These gate cuts were initially planned in the mine permit, but the original plans were not sufficient to mitigate the extensive pooling that resulted following subsidence. A permit revision was therefore required to amend the restoration plans. The Muddy Creek gate cut in Emerald Mine is also awaiting approval by PADEP. All other gate cuts listed in Table VII-12 have been approved and await action by the mine operator.

Table VII-12. List of all stream segments with proposed gate cuts during the 4th assessment. Gate cuts had not been initiated as of 20 August 2013.

Mine	PA WRDS Stream Code	Stream Name	Panel
Bailey	32551	Mudlick Fork	15I
Bailey	32551	Mudlick Fork	16I
Blacksville 2	41812	Blockhouse Run	13-14W
Blacksville 2	41812	Blockhouse Run	13-14W
Blacksville 2	41812	Blockhouse Run	14-15W
Blacksville 2	41812	Blockhouse Run	15-16W
Blacksville 2	41821	UNT to Blockhouse Run	17W
Blacksville 2	41821	UNT to Blockhouse Run	18W
Emerald	41268	Mount Phoebe Run	B6
Emerald	41014	Muddy Creek	C2
Emerald	41246	Dutch Run	B7
Enlow Fork	32777	Buffalo Creek	F22
Enlow Fork	40285	Ten Mile Creek	E23
Enlow Fork	40285	Ten Mile Creek	E24

VII.I.3 – Augmentation, grouting, and stream liner methods

When a flow loss impact occurs following mining, augmentation is used to restore flow to the stream within 24 hours (if flow loss was predicted) or within 15 days (if flow loss was unpredicted; PADEP 2005a). Augmentation restores flow by drawing on water from nearby wells, water tanks, or from other streams. The water is carried above and belowground via a system of pipelines from these sources to the augmentation discharge points (Figure VII-14). Depending on the severity of the impact, multiple augmentation points may be necessary to sustain stream flow across the impacted area. At the discharge point, the water is either released directly into the stream or released into a rock lined valley that was constructed along the stream bank. The rock valley aerates the water, which can remove certain pollutants, and prevents bank erosion. It has been suggested that the augmentation process may wash alluvial material into small surface cracks and allow the stream to "self-heal". However, the University could not identify any clear cases of "self-healing", suggesting that for many streams, additional mitigation work is required to repair the flow loss impacts.

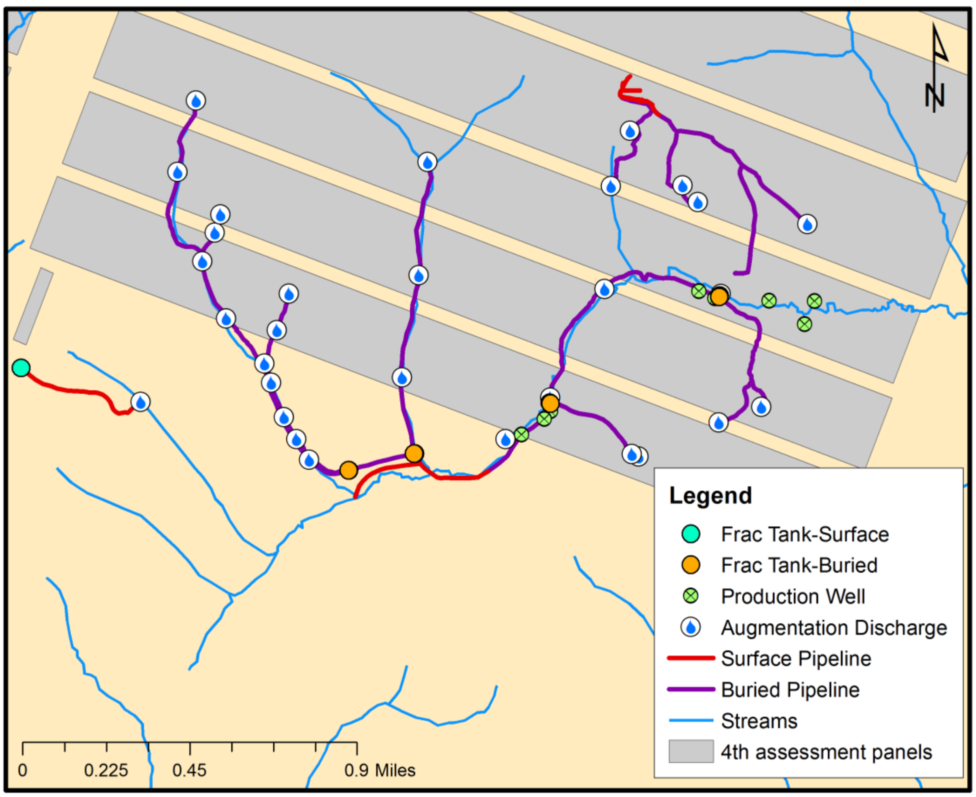

Figure VII-14. The augmentation system in the Crafts Creek watershed over Enlow Fork Mine. It is one of the larger augmentation systems developed by mine operators during the 4th assessment period. Map does not show zero-order tributaries, many of which also received augmentation (see right side of map).

To mitigate flow loss impacts, mine operators typically turn to a technique known as grouting. Depending on the severity of the impact, one of two grouting methods may be applied – surficial grouting or closure grouting (Haibach et al. 2012). For both methods, a pump-around system is set up to divert stream flow and dry the stream bed. Bedrock heaves and excess alluvial material are then removed. For impacts requiring surficial grouting, fractures in the bedrock are identified and sealed using a cement mixture containing roughly 3% bentonite clay. The clay provides flexibility to the cement mixture and prevents the cement from shrinking once it hardens in the cracks (B. Benson, Consol Energy, Inc., pers. comm.). Once the surface cracks are filled, the stream banks are stabilized and in-streams habitats are restored (Haibach et al. 2012). If surficial grouting is not successful in restoring flow, then closure grouting may be used. In closure grouting, a series of 2-in diameter, 6-ft deep boreholes are drilled at 10-ft intervals across the impacted area. The cement/bentonite clay mixture's viscosity is finely tuned to a thin watery paste, and pumped into the boreholes at a low steady velocity that is designed to fill in even very small cracks in the underlying bedrock. If this first "pass" is not successful in restoring stream flow, then a second pass will drill and fill boreholes at 5-ft intervals across the impacted area. Eventually, the rock fissures will not take any more grout, and grouting mitigation will cease.

While grouting is the preferred method for mitigating flow loss on streams with bedrock bottoms, it is ineffective on streams where the alluvial thickness is greater than 3-ft (Haibach et al. 2012). In these cases, the mine operator may opt to install a channel lining to seal the cracks. Channel lining is also used as a last resort on bedrock dominated streams when grouting has proven to be ineffective. Channel linings come in two forms – a synthetic liner and an alluvial amendment (Haibach et al. 2012). While channel linings have been in use for several years (e.g. the Laurel Run channel lining installed in 2007 above Emerald Mine), the technology for this mitigation method is continually evolving. In the past, channel linings consisted exclusively of synthetic liners. Here, the most recent innovations in channel linings are described.

To install a channel lining, the vegetation surrounding the construction site is first cleared and the necessary erosion and sedimentation controls are set up. The alluvium in the stream channel is then excavated and stockpiled nearby. For synthetic liners, a geosynthetic clay fabric is laid across the stream channel and the fabric is topped with a cellular confinement system (Figure VII-15a). This honeycomb-shaped system of cells is anchored into the ground and filled with gravel (Figure VII-15b). Once the stream enhancement structures are in place, the cells are topped with a thick layer (i.e. minimum of 6-in) of the stockpiled alluvium.

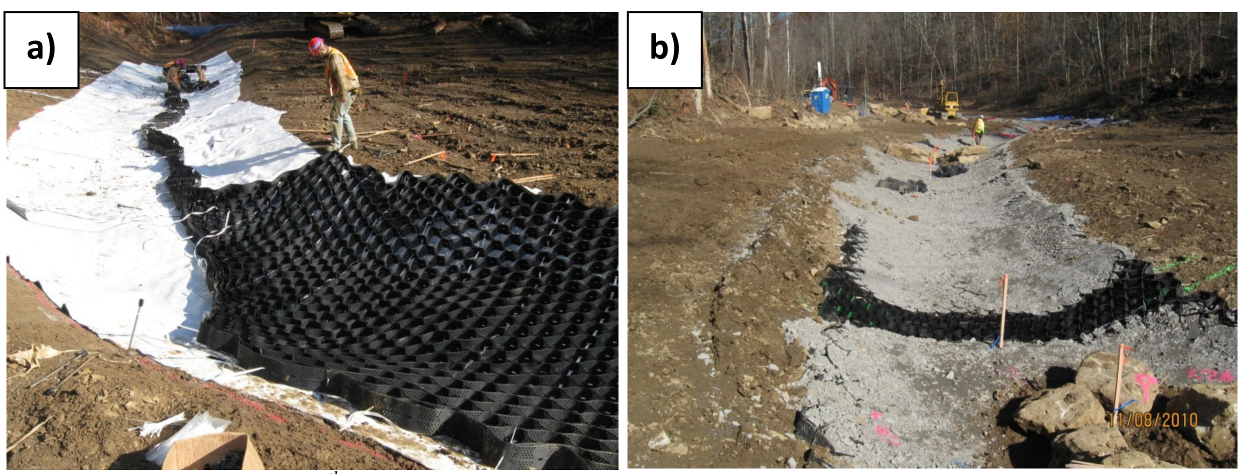

Figure VII-15. During the 4th assessment the installation of a synthetic liner consisted of a) a geosynthetic clay liner and a cellular confinement system and b) gravel fill (Photo from PADEP files).

In contrast, for an alluvial amendment, the stockpiled alluvium is mixed with bentonite clay. Typically, 100-ft^3 of alluvium and soil material are mixed with 200-lbs of bentonite to form a slurry (B. Benson, Consol Energy, Inc., pers. comm.). The slurry is laid down in the excavated stream channel and compacted to create a channel lining (Figure VII-16). Following installation of either the synthetic liner or alluvial amendment, the stream banks are re-graded, stabilized, and planted in a similar fashion to a gate cut mitigation project.

Figure VII-16. Placement of alluvial amendment liner into stream channel in E18 panel of Crafts Creek (Photo from PADEP files).

VII.I.4 – Augmentation, grouting, and stream liners during the 4th assessment

To determine the number of streams receiving augmentation during the 4th assessment period, the University collected data from the PADEP monthly flow map database. For mines operated by Consol Energy, Inc., Microsoft Excel files describe both the number of augmentation discharge points that are installed on a stream (i.e. available for use) and the number of augmentation points that are active on a stream in a particular month. The University compiled data for all months and used SAS (SAS Institute Inc. 2013) to identify the maximum number of discharges installed on each stream and the maximum number that were active at one time. Unfortunately, for Enlow Fork and Blacksville 2 Mines, data on augmentation was not submitted to the PADEP monthly flow map database prior to June 2011. Thus, the analysis of total augmentation in these mines is conservative and only represents augmentation that occurred from June 2011 until the end of the reporting period. For mines operated by Alpha Natural Resources, the University used the “straight-line” maps to identify streams receiving augmentation. Unfortunately, the “straight-line” maps do not provide information on the number of augmentation discharges that are installed or active on a stream and the University could locate this information in any of the files at CDMO.

Augmentation discharge points were installed on 95 streams during the 4th assessment period (Table VII-13). Augmentation discharges were active at 74 of these streams (Table VII-13).

Table VII-13. Streams receiving augmentation during the 4th assessment period.

Mine	PA WRDS Stream Code[1]	Stream Name	Company Code	Max # of Augmentation Discharges Installed	Max # of Augmentation Discharges Active At One Time
Bailey	32540	Barneys Run	BarR	6	5
Bailey	NA	UNT to Barneys Run	BarR-2R	3	1
Bailey	NA	UNT to UNT of Barneys Run	BarR-2R-1L	1	1
Bailey	32541	UNT to Barneys Run	BarR-3R	2	2
Bailey	32542	UNT to Barneys Run	BarR-5R	4	3
Bailey	NA	UNT to UNT of Barneys Run	BarR-5R-4R	1	1
Bailey	32544	UNT to Barneys Run	BarR-8R	3	4
Bailey	NA	UNT to UNT of Barneys Run	BarR-8R-1R	3	2
Bailey	32543	UNT to Barneys Run	BarR-8R_BarR-8R-2R	3	3
Bailey	32532	UNT to Dunkard Fork	DF-19R	10	4
Bailey	32533	UNT to UNT of Dunkard Fork	DF-19R-1R	3	2
Bailey	32534	UNT to UNT of Dunkard Fork	DF-19R-2R	3	3
Bailey	NA	UNT to UNT of Dunkard Fork	DF-19R-2R-1R	3	3
Bailey	32535	UNT to UNT of Dunkard Fork	DF-19R-5R	1	0
Bailey	NA	UNT to UNT of Dunkard Fork	DF-19R-7R	2	2
Bailey	32511	UNT to Dunkard Fork	DF-9L	3	3
Bailey	32553	UNT to Hewitt Run	HewR-2R	3	3
Bailey	32551	Mudlick Fork	MdlkF	2	2
Bailey	NA	Crow's Nest	NoF-1L	3	2
Bailey	32595	UNT to North Fork of Dunkard Fork	NoF-3L	5	4
Bailey	32597	UNT to North Fork of Dunkard Fork	NoF-3R	2	1
Bailey	32596	UNT to North Fork of Dunkard Fork	NoF-5L	10	7
Bailey	32598	Polly Hollow	PlyH	1	1
Bailey	32536	South Fork of Dunkard Fork	SoF	22	10
Bailey	NA	UNT to South Fork of Dunkard Fork	SoF-11L	4	3

Mine	PA WRDS Stream Code[1]	Stream Name	Company Code	Max # of Augmentation Discharges Installed	Max # of Augmentation Discharges Active At One Time
Bailey	32566	UNT to South Fork of Dunkard Fork	SoF-12L	3	3
Bailey	32567	UNT to South Fork of Dunkard Fork	SoF-16R	2	0
Bailey	32537	UNT to South Fork of Dunkard Fork	SoF-1R	1	1
Bailey	NA	UNT to South Fork of Dunkard Fork	SoF-2R	2	1
Bailey	NA	UNT to South Fork of Dunkard Fork	SoF-3R	1	0
Bailey	NA	UNT to South Fork of Dunkard Fork	SoF-4R	1	1
Bailey	32539	UNT to South Fork of Dunkard Fork	SoF-5L	5	4
Bailey	32546	UNT to South Fork of Dunkard Fork	SoF-6L	5	3
Bailey	32549	UNT to South Fork of Dunkard Fork	SoF-8L	4	3
Bailey	32550	UNT to UNT of South Fork of Dunkard Fork	SoF-8L-2L	2	2
Bailey	32565	UNT to South Fork of Dunkard Fork	SoF-9L	7	3
Bailey	32547	Strawn Hollow	StrnH	4	3
Bailey	NA	UNT to Strawn Hollow	StrnH-1R	1	0
Bailey	32548	UNT to Strawn Hollow	StrnH-2R	3	2
Bailey	32504	Wharton Run	WhrtnR	3	2
Bailey	NA	UNT to Wharton Run	WhrtnR-7L	2	1
Bailey	32508	UNT to Wharton Run	WhrtnR-8R	1	1
Blacksville 2	41812	Blockhouse Run	BlkhR	8	6
Blacksville 2	41826	UNT to Blockhouse Run	BlkhR-15R	4	3
Blacksville 2	41820	UNT to Blockhouse Run	BlkhR-1L	3	3
Blacksville 2	41824	UNT to Blockhouse Run	BlkhR-2L	2	2
Blacksville 2	41818	UNT to Blockhouse Run	BlkhR-2R	4	4
Blacksville 2	41819	UNT to Blockhouse Run	BlkhR-3R	3	3
Blacksville 2	41821	UNT to Blockhouse Run	BlkhR-9R	1	0
Blacksville 2	41813	Roberts Run	RbtsR	1	1

Mine	PA WRDS Stream Code[1]	Stream Name	Company Code	Max # of Augmentation Discharges Installed	Max # of Augmentation Discharges Active At One Time
Blacksville 2	41806	Toms Run	TmsR	1	0
Blacksville 2	41809	UNT to Toms Run	TmsR-4R	1	0
Blacksville 2	41833	UNT to Toms Run	TmsR-8R	1	1
Cumberland	41282	UNT to Whiteley Creek	WC_41282	No data	No data
Cumberland	40614	UNT to Pursley Creek	PC_40614	No data	No data
Cumberland	NA	UNT to Pursley Creek	PC_40592-L7	No data	No data
Cumberland	41264	UNT to Dyers Fork	DF_41264	No data	No data
Cumberland	41267	UNT to Dyers Fork	DF_41267	No data	No data
Emerald	41014	Muddy Creek	MC_41014	No data	No data
Emerald	41252	UNT to Dutch Run	DR_41252	No data	No data
Enlow Fork	32777	Buffalo Creek	BufC	5	0
Enlow Fork	32998	UNT to Buffalo Creek	BufC-11R	2	0
Enlow Fork	32999	UNT to Buffalo Creek	BufC-12R	3	2
Enlow Fork	33000	UNT to Buffalo Creek	BufC-13R	1	1
Enlow Fork	32996	UNT to Buffalo Creek	BufC-9L	2	2
Enlow Fork	40938	Crafts Creek	CrC	10	5
Enlow Fork	NA	UNT to Crafts Creek	CrC-1.5R	1	0
Enlow Fork	NA	UNT to Crafts Creek	CrC-1.7R	2	2
Enlow Fork	40939	UNT to Crafts Creek	CrC-1R	2	2
Enlow Fork	40940	UNT to UNT of Crafts Creek	CrC-1R,2R	3	2
Enlow Fork	40941	UNT to Crafts Creek	CrC-2R	2	2
Enlow Fork	40942	UNT to Crafts Creek	CrC-3R	5	4
Enlow Fork	40943	UNT to UNT of Crafts Creek	CrC-3R,1R	2	1
Enlow Fork	40944	UNT to Crafts Creek	CrC-4R	10	5
Enlow Fork	NA	UNT to UNT of Crafts Creek	CrC-4R,2R	2	2

Mine	PA WRDS Stream Code[1]	Stream Name	Company Code	Max # of Augmentation Discharges Installed	Max # of Augmentation Discharges Active At One Time
Enlow Fork	NA	UNT to UNT of Crafts Creek	CrC-4R,3R	2	0
Enlow Fork	40945	UNT to Crafts Creek	CrC-5R	1	0
Enlow Fork	NA	UNT to Crafts Creek	CrC-6L	2	0
Enlow Fork	NA	UNT to UNT of Crafts Creek	CrC-6L,1L	1	0
Enlow Fork	NA	UNT to Crafts Creek	CrC-9L	2	0
Enlow Fork	32719	UNT to Rocky Run	RkyR-9L	5	0
Enlow Fork	32708	Templeton Fork	TemF	6	3
Enlow Fork	32738	UNT to Templeton Fork	TemF-21L	1	0
Enlow Fork	NA	UNT to UNT of Templeton Fork	TemF-21L,0.9L	2	0
Enlow Fork	32739	UNT to UNT of Templeton Fork	TemF-21L,1L	1	0
Enlow Fork	32740	UNT to Templeton Fork	TemF-23L	6	0
Enlow Fork	32742	UNT to Templeton Fork	TemF-25L	3	1
Enlow Fork	NA	UNT to UNT of Templeton Fork	TemF-25L,1L	3	0
Enlow Fork	32743	UNT to Templeton Fork	TemF-26L	2	1
Enlow Fork	32744	UNT to Templeton Fork	TemF-27L	2	2
Enlow Fork	32745	UNT to Templeton Fork	TemF-28L	2	1
Enlow Fork	40285	Ten Mile Creek	TenC	2	1
Enlow Fork	40949	UNT to Ten Mile Creek	TenC-8L	3	2
Enlow Fork	40951	UNT to UNT of Ten Mile Creek	TenC-8L,1L	3	3
Enlow Fork	40950	UNT to UNT of Ten Mile Creek	TenC-8L,2R	2	2

[1]NA = Zero order tributary

Bailey Mine had the greatest number of streams with installed and active augmentation discharge points (Figure VII-17). While Enlow Fork Mine undermined more miles of stream than Bailey Mine (Table VII-5), this mine installed fewer augmentation discharges and activated only half as many discharges (Figure VII-17). The analysis accounts for augmentation on streams that currently have an augmentation reprieve from PADEP so reprieves do not explain the differences between the two mines. The University expects that the differences are the result of either fewer flow loss impacts in Enlow Fork Mine or an artifact of the incomplete dataset for Enlow Fork. What is clear from Figure VII-17 is that the number of streams receiving active augmentation in Bailey Mine was nearly 5-times that of Blacksville 2 Mine, 8-times that of Cumberland Mine, and nearly 20-times that of Emerald Mine. These differences are due in part to the fact that these mines undermined fewer miles of stream than Bailey Mine (Table VII-5). The differences may also reflect variation in the mine operators' approaches to stream mitigation and/or dissimilarities in the geologic and hydrologic conditions between the mines.

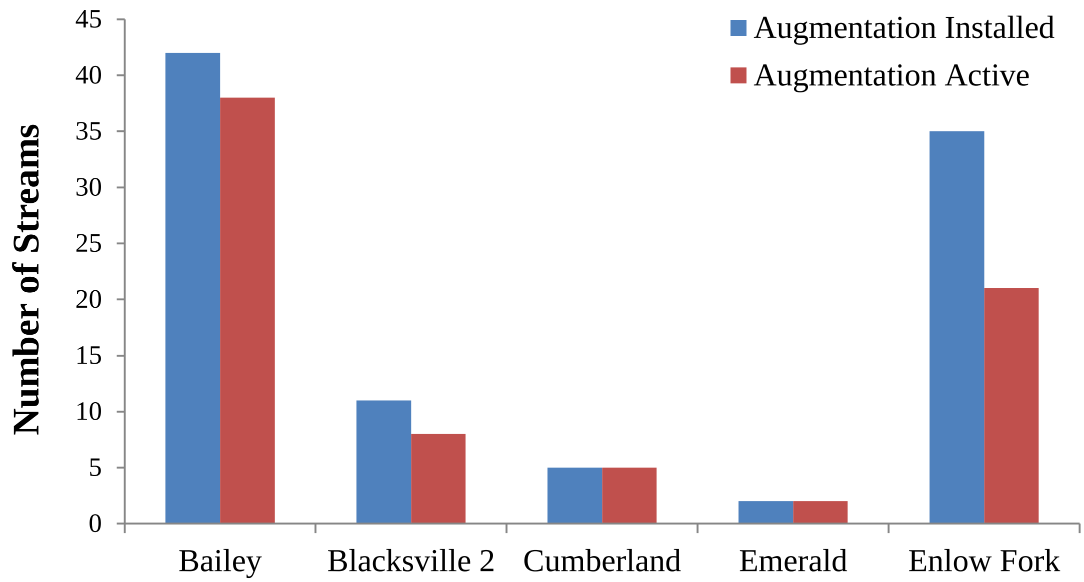

Figure VII-17. Number of streams with augmentation installed and active during the 4th assessment period by mine.

On average, a stream receiving augmentation had 3.20 +/- 0.31 augmentation discharges installed along its length, but only a maximum of 1.94 +/- 0.19 discharges were active at any one time (mean +/- 1 standard error). South Fork of Dunkard Fork in Bailey Mine had the greatest number of augmentation points installed with 22 discharges along its length (Table VII-13). During the drought period of August 2010, up to 10 augmentation points were actively discharging over 600-gpm into South Fork of Dunkard Fork. Other streams with significant numbers of augmentation points installed include: an unnamed tributary to Dunkard Fork (Stream 32532), an unnamed tributary to the North Fork of Dunkard Fork (Stream 32596), Crafts Creek, and an unnamed tributary to Crafts Creek (Stream 40944) (Table VII-13).

To determine the number of streams receiving grouting during the 4th assessment period, the University compiled data from two major sources: the BUMIS agent observation files and the

SSA stream data logs (see Section VII.F for description). The extent of grouting on each stream was approximated by recording the panels in which grouting occurred.

The University found that 57 streams received grouting during the 4th assessment period (Table VII-14). Of these, 40% received grouting in multiple panels. PADEP does not currently require mine operators to report the length of stream grouted, but the University suggests that these data would be useful in assessing the actual extent of stream mitigation following mining. The University was able to locate one report describing stream grouting in Bailey Mine for the 3rd and 4th quarters of 2008 (Consol Energy Inc. 2009). The report was in a folder labelled "Bailey Mine Special Conditions" in the CDMO paper files. According to the report, ~5,941-ft and ~2,758-ft of streams were grouted in the 3rd and 4th quarters of 2008, respectively. It is unclear from the report if the amount of grouting in these two quarters is representative of the grouting extents in a typical year. However, if these lengths are extrapolated across the five-year reporting period, it would be predicted that over 8 miles of stream received grouting during the summers and falls of 2008-2013 in Bailey Mine. Considering that 16.75 miles of stream were undermined by Bailey Mine during this assessment period (Table VII-1), if this estimate of the grouting extent is even reasonably close, then ~50% of the stream length undermined in Bailey Mine was likely grouted. The University suspects that this estimate of grouting in Bailey is highly conservative. First, the University does not have an estimate of grouted stream lengths for the 1st and 2nd quarters of 2008, so the amount of stream that may have been grouted during the winter and spring months of the reporting period is unknown. Second, the report indicates that between December 2006 and September 2008, ~35,935-ft of stream were grouted in Bailey Mine (Consol Energy Inc., 2009). This indicates that in less than two years, nearly 7 miles of stream received grouting, and suggests that the estimate of 8 miles of grouting over a 5 year period is likely too low. Unfortunately, the University could not locate similar data from other mines for comparison with this analysis on Bailey Mine.

Overall, these data suggest that bedrock fracturing is a widespread accompaniment to longwall mining, at least in Bailey Mine. Bedrock fracturing is likely a common feature of undermined landscapes in areas with geological profiles that are similar to those in Bailey Mine. The extent of bedrock fracturing in such mines, and its influence on shallow groundwater flow, necessitates an extensive grouting mitigation program, perhaps to a much greater degree than has previously been recognized.

Table VII-14. Streams receiving grouting in the 4th assessment period.

Mine	**PA WRDS Stream Code[1]**	**Stream Name**	**Company Code**	**Panels with Grouting**
Bailey	32540	Barneys Run	BarR	11I, 12I, 16H, 17H
Bailey	NA	UNT to Barneys Run	BarR-2R	11I
Bailey	32542	UNT to Barneys Run	BarR-5R	14H
Bailey	32544	UNT to Barneys Run	BarR-8R	16H
Bailey	32543	UNT to Barneys Run	BarR-8R and BarR-8R-2R	16H

Mine	PA WRDS Stream Code[1]	Stream Name	Company Code	Panels with Grouting
Bailey	NA	UNT to Barneys Run	BarR-8R-1R	15H
Bailey	32532	UNT to Dunkard Fork	DF-19R	10H, 11H, 12H, 13H
Bailey	NA	UNT to Dunkard Fork	DF-19R-7R	13H
Bailey	32511	UNT to Dunkard Fork	DF-9L	16C
Bailey	32553	UNT to Hewitt Run	HewR-2R	15I, 16I
Bailey	NA	UNT to Mudlick Fork	MdlkF-1L	16I
Bailey	NA	Crow's Nest	NoF-1L	5I
Bailey	32595	UNT to North Fork of Dunkard Fork	NoF-3L	2I, 3I, 4I
Bailey	32596	UNT to North Fork of Dunkard Fork	NoF-5L	2I, 4I
Bailey	32598	Polly Hollow	PlyH	4I
Bailey	32536	South Fork of Dunkard Fork	SoF	8I, 12I
Bailey	NA	UNT to South Fork of Dunkard Fork	SoF-11L	14I
Bailey	32566	UNT to South Fork of Dunkard Fork	SoF-12L	14I, 15I
Bailey	32567	UNT to South Fork of Dunkard Fork	SoF-16R	15I
Bailey	32539	UNT to South Fork of Dunkard Fork	SoF-5L	10I, 11I
Bailey	32546	UNT to South Fork of Dunkard Fork	SoF-6L	11I
Bailey	32549	UNT to South Fork of Dunkard Fork	SoF-8L	10I, 11I, 12I
Bailey	32550	UNT to South Fork of Dunkard Fork	SoF-8L-2L	10I, 11I
Bailey	32565	UNT to South Fork of Dunkard Fork	SoF-9L	12I, 13I
Bailey	32547	Strawn Hollow	StrnH	14I
Bailey	32548	UNT to Strawn Hollow	StrnH-2R	13I
Blacksville 2	41812	Blockhouse Run	BlkhR	15W, 16W
Blacksville 2	41826	UNT to Blockhouse Run	BlkhR-15R	18W
Blacksville 2	41820	UNT to Blockhouse Run	BlkhR-1L	15W
Blacksville 2	41818	UNT to Blockhouse Run	BlkhR-2R	14W, 15W
Blacksville 2	41819	UNT to Blockhouse Run	BlkhR-3R	15W, 16W, 17W
Blacksville 2	41813	Roberts Run	RbtsR	14W
Blacksville 2	41833	UNT to Tom's Run	TmsR-8R	21M
Cumberland	41261	Dyers Fork	DF_41261	52, 53, 54
Cumberland	41264	UNT to Dyers Fork	DF_41264	50, 51, 52, 53
Cumberland	41267	UNT to Dyers Fork	DF_41267	51
Cumberland	41246	Dutch Run	DR_41246	51-52 gate, 53
Cumberland	NA	UNT to Pursley Creek	PC_40592-L7	60
Cumberland	40614	UNT to Pursley Creek	PC_40614	60
Cumberland	NA	UNT to Turkey Hollow	TH_40611-L2	60

Mine	PA WRDS Stream Code[1]	Stream Name	Company Code	Panels with Grouting
Cumberland	41282	UNT to Whiteley Creek	WC_41282	55
Emerald	41252	UNT to Dutch Run	DR_41252	B6
Emerald	41014	Muddy Creek	MC_41014	C1
Emerald	40465	UNT to Smith Creek	SC_40465	E2
Enlow Fork	40938	Crafts Creek	CrC	E18, E19, E20
Enlow Fork	NA	UNT to Crafts Creek	CrC-1.5R	E20
Enlow Fork	NA	UNT to Crafts Creek	CrC-1.7R	E20
Enlow Fork	40941	UNT to Crafts Creek	CrC-2R	E20
Enlow Fork	NA	UNT to Crafts Creek	CrC-3L	E20
Enlow Fork	40942	UNT to Crafts Creek	CrC-3R	E20
Enlow Fork	40943	UNT to Crafts Creek	CrC-3R,1R	E20, E21
Enlow Fork	40944	UNT to Crafts Creek	CrC-4R	E17
Enlow Fork	32742	UNT to Templeton Fork	TemF-25L	F17, F18
Enlow Fork	32744	UNT to Templeton Fork	TemF-27L	F18, F19
Enlow Fork	40949	UNT to Ten Mile Creek	TenC-8L	E22, E23
Enlow Fork	40951	UNT to Ten Mile Creek	TenC-8L,1L	E22, E23
Enlow Fork	NA	UNT to Templeton Fork	TF21L-0.5L	F16

[1]NA = Zero order tributary

Lastly, the University investigated the number of stream impacts requiring liners and the time to restoration. Based on data from the permit revisions and conversations with the SSAs and mine operators, the University found that three streams had liners installed during the 4th assessment period (Table VII-15). The time to restoration for each liner project was calculated as the date the liner project began minus the date of undermining. Dates were determined from SSA observation records in BUMIS. Details regarding the liner installations are provided below.

Table VII-15. Streams with liners installed during the 4th assessment period.

Mine	PA WRDS Stream Code	Stream Name	Panel	Time to Restoration (days)	Liner Type: Length (ft)
Bailey	32596	UNT to North Fork of Dunkard Fork	3I	2074	synthetic: 1,150
Enlow Fork	40938	Crafts Creek	E18	674	synthetic: 607 alluvial: 450
Mine 84	40824	Brush Run	6B	2673	alluvial: 750

In Bailey Mine, Stream 32596 (an unnamed tributary to North Fork of Dunkard Fork) exhibited extensive flow loss following subsidence of the 1-4I panels. The flow loss in the 3I panel was tracked by the PADEP in stream investigation ST0502 and ST1203 (see Section VIII.B.4). A total of 10 augmentation points were installed along this stream to maintain flow (Table VII-13) and by the end of December 2008, nearly 75% of the stream length had been grouted (Consol Energy Inc. 2009). Despite these mitigation efforts, stream flow did not recover in the 3I panel and the mine operator began working with property owners as early as July 2009 to obtain access for additional restoration work, such as liner installation or additional grouting (Consol Pennsylvania Coal Company, LLC 2009). Installation of an 1,150-ft strip of synthetic liner occurred during the fall of 2010 over the 3I panel. The following spring the liner experienced damage during high flow events. The SSA noted that water had seeped in underneath the synthetic liner, the cellular confinement system had become exposed, and the banks were eroding in some places. BUMIS records indicate that liner repair did not begin until 20 August 2012, over a year after the damages occurred. Observations from the fall of 2013 noted that the cellular confinement system was still visible in places but that no new bank erosion had occurred. Following a period of monitoring, PADEP ultimately ruled that this stream had not been restored to its pre-mining condition in the 3I panel or any other panel (ST1203).

Crafts Creek was impacted by flow loss in the E18 panel almost immediately after longwall mining passed under the stream. BUMIS records indicate that on 12 November 2008 a “during mining” survey of the E18 panel revealed a 1,400-ft section of flow loss and ~200 dead fish. PADEP ordered immediate augmentation and issued a compliance order as a result of the infraction. The augmentation system in the Crafts Creek watershed is extensive (Figure VII-14) and was used to maintain flow until restoration work could begin. Grouting approaches were not effective in sealing the fractures beneath the stream. PADEP approved a permit revision for the installation of a 1,050-ft liner on 3 August 2010. Rather than line the entire restoration area with a synthetic liner, the operator split the restoration area in half to compare the effectiveness of a synthetic liner against the new alluvial amendment approach. Restoration work was completed in the fall of 2010. However, as with the liner for stream 32596, the SSA noted damage to the synthetic liner the following spring. The cellular confinement system was exposed and groundwater was pushing up from beneath the stream channel against the liner, causing the liner to balloon up in the stream. No significant problems were noted in the area of the alluvial amendment. During the fall of 2011, French drains were installed to relieve the groundwater pressure around the synthetic liner. The drains discharge directly into Crafts Creek. When the University observed the restoration area in July 2013, the synthetic liner and cellular confinement systems were not exposed, suggesting that the French drains have been effective in allowing the liner to settle. Recovery of the biological community over the E18 liner has not yet been assessed. However, biological data collected following the E18 gate cut indicates that the Total Biological Score at the bio-monitoring station in the E18 panel (BSW16) has returned to pre-mining levels (pre-mining average TBS: 49.4 vs. a single post-gate cut TBS: 45.8; CEC 2010). It should be noted that the pre-mining scores for this site did not meet the requirements of TGD 563-2000-655 – the relative percent difference among the pre-mining scores was greater than 16%.

The last liner installed during the 4th assessment was on Brush Run in Mine 84. This stream was undermined by the 6B panel between 20 March and 11 April 2006. BUMIS records indicate that on 26 April 2006, the stream was pooling near the 6B-7B gate but was dry in the center of the 6B panel. The 6B-7B gate was cut in March 2007 to alleviate the pooling, however, the flow loss problems remained. The shallow overburden at this site (380-ft) coupled with the unique location of the stream on the edge of the panel may have contributed to the flow loss. At least three augmentation wells were drilled in panel 6B to sustain flow over the impacted area. The mine operator began negotiating with surrounding landowners in June 2009 to gain access to the stream for restoration. An agreement between the operator and landowner was reached in May 2011. It was decided that grout mitigation would be ineffective in Brush Run because the bedload is greater than 6-ft deep in some places. The mine operator proposed to utilize either a synthetic liner or alluvial amendment to mitigate the flow loss. Once the access agreement was in place, the operator requested an extension from PADEP to wait to begin restoration work until spring 2012. During the summer of 2012, the operator applied for and received an Army Corps of Engineers permit for the stream work. The operator opted to use an alluvial amendment on Brush Run, likely due in part to the reduced number of problems associated with this type of liner installation at Crafts Creek. Restoration work in the Brush Run stream channel finally began on 5 August 2013. Overall, the time to restoration for this stream segment was >7 years. While obtaining landowner access clearly played a part in delaying the mitigation process, it is unclear why the operator did not immediately apply for the required permits after gaining access. It is also interesting to note that this flow loss did not trigger a stream investigation during the 3rd Act 54 assessment period at the PADEP, and as result, no information was reported on this impact in the last Act 54 report (Iannacchione et al. 2011). Unfortunately, the effectiveness of this liner installation could not be evaluated by the University because the mitigation work occurred so close to the end of the current assessment period. The University suggests that future studies follow up on this stream restoration project.

VII.I.5 – Construction of Access Roads to Support Mitigation Activities

Many undermined streams are surrounded by forest or other habitats that are difficult to navigate with the heavy construction equipment that is required for mitigation. As a result, mine operators build roads along impacted streams to facilitate the movement of equipment between existing roads and the mitigation site. Road construction is an ecological disturbance that is generally associated with declines in biodiversity in both terrestrial and aquatic communities (Trombulak and Frissell 2000). Roads affect communities through multiple mechanisms, including vegetation removal during construction, modification of animal behavior, alteration of both the physical and chemical environment, spread of invasive species, and increasing the use of the area by humans (Trombulak and Frissell 2000). While road construction is not a direct result of subsidence, it is an indirect effect of mitigating subsidence-related stream impacts. To fully understand the impacts of subsidence and subsequent restoration on stream ecosystems, it is important to consider the effect of road construction.

VII.I.6 – Access Road Construction during the 4th Assessment

While access road construction can be quantified through careful, time-consuming measurements of the erosion and sedimentation control plans submitted to PADEP, mine operators are not required to formally report this information in the mitigation plans. However, the University was able to locate one report for Bailey Mine that detailed access road construction during the 3rd and 4th quarters of 2008 (Consol Energy Inc. 2009). No access roads were built during the 3rd quarter of 2008. However, ~7,913-ft of access roads were built in the 4th quarter (Consol Energy Inc.

2009). Overall, the report states that between December 2006 and December 2008, access road construction in Bailey Mine totaled 79,426-ft, or 15 miles (Figure VII-18; Consol Energy Inc. 2009). Because data from other time periods and other mines was not available, it is unknown if this amount of road construction is representative of conditions at all longwall mines. The University suggests that PADEP request that mine operators specifically quantify and report the length of access road construction as this would provide valuable information regarding the degree of disturbance to terrestrial and aquatic ecosystems during mitigation.

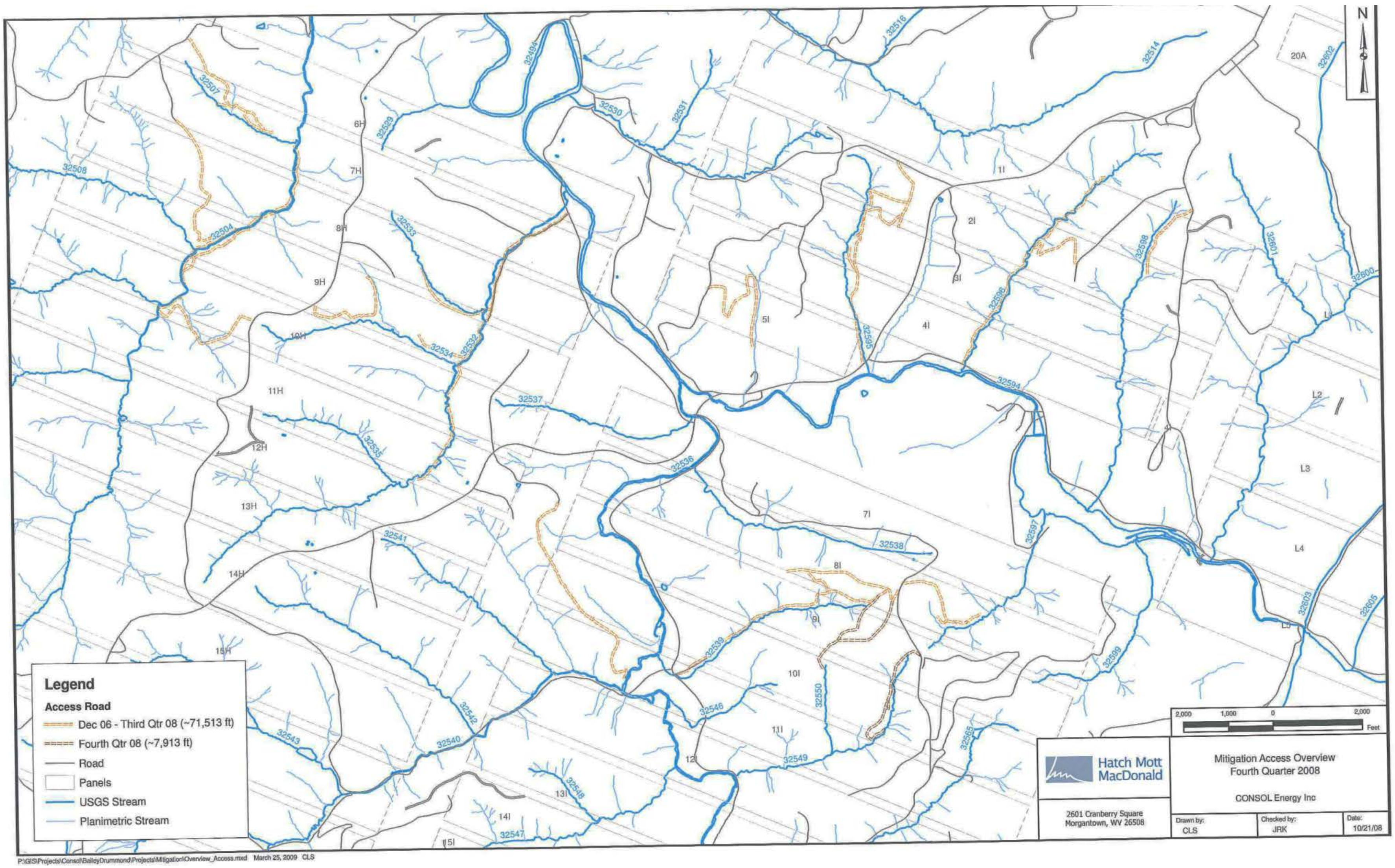

Figure VII-18. Map from mine operator showing access road construction (orange lines) in Bailey Mine between December 2006 and December 2008 (Consol Energy Inc. 2009).

VII.J – Biological Assessment of Streams Following Mitigation

While stream macroinvertebrates communities are affected by mining-induced flow loss and pooling, it is unknown if the mitigation measures (i.e. augmentation, grouting, liners, gate cuts) utilized by mining companies are effective in restoring the communities. PADEP tasked the University with evaluating at least five stream segments that had gate cut mitigation work completed during the 4th assessment period. While the University was not specifically tasked with evaluating stream segments that had grout mitigation, it is clear that far more stream segments receive grouting than gate cuts (Table VII-14 vs. Table VII-13). Thus, the University was also interested in determining the biological recovery of streams that receive grouting.

VII.J.1 – Stream Biology Data Collection and Analysis

The University could not identify any TBS that were specifically identified as being collected “post-grouting”. The TBS collected after mining at sites that are known to have received grouting are identified as simply “post-mining”. Because the date of grouting is unknown, it is uncertain if these “post-mining” TBS were collected before or after grout mitigation. It is possible that some of the post-mining TBS data in the flow loss analysis (Figure VII-7) were collected after grouting took place. The University suggests that PADEP require mine operators to label samples collected after grout mitigation as “post-grouting” as this would allow for PADEP to assess the effectiveness of this mitigation technique.

While the University could not look at the effect of grouting per se, it was possible to investigate changes in TBS and stream chemistry over time at sites experiencing flow loss. The University predicted that if grouting is effective in restoring stream biology and chemistry to pre-mining levels, then TBS would increase and pH and conductivity would decrease with time since mining. To test this prediction, the University used the flow loss dataset described above (see Section VII.H.1). To determine the time since mining, the University first geo-referenced the monthly longwall face positions for Bailey Mine to estimate the date when each bio-monitoring station was undermined. The date of undermining was considered to be the date at which the face had passed the station. For all post-mining biological and chemical samples, the time since mining was then calculated by subtracting the date of undermining from the date of sampling. Regression analyses were used to test the relationship between time since mining and TBS (raw and adjusted), conductivity, and pH. Analyses controlled for the effects of station on biology and chemistry.

For streams with gate cut mitigation, pre-mining and post-restoration TBS are located in the annual stream restoration reports that are submitted to PADEP. The University identified a total of 18 bio-monitoring stations with post-restoration biology data. The stations are located near gate cuts in Bailey Mine, Enlow Fork Mine, and Cumberland Mine. The three stations in Cumberland Mine however, are located ~1200-ft upstream and/or downstream from the actual restoration areas. Samples from these stations may not reflect the conditions of the biological community with the actual restoration area. It is unclear why bio-monitoring stations were not established inside the gate cuts at Cumberland Mine. Therefore, data from Cumberland Mine are presented separately. For the data from Bailey and Enlow Fork Mine, bio-monitoring stations were located directly within the restoration area. The data were corrected for spatial autocorrelation by eliminating stations that were within 1,673-ft of another station (see Section VII.C.2 for methodology). Correction for temporal correlation was not possible due to the timing of the sampling events. The corrected dataset contained pre-mining and post-restoration TBS data from 10 bio-monitoring stations (N = 67 samples). The University tested for the difference between pre-mining and post-restoration raw and adjusted TBS using an ANOVA (model: raw or adjusted TBS = mining (pre-mining vs. post-restoration) + station). Adjusted TBS were corrected for the effects of surrounding land use and month of sampling (as described in Sections VII.C.2 and VII.C.3).

VII.J.2 – Relationship between Time since Mining and Total Biological Score, Conductivity, and pH for Streams Impacted by Mining-Induced Flow Loss

On average, TBS collected at bio-monitoring stations with mining-induced flow loss increase over time (effect of time since mining on raw TBS: $F_{1,51} = 10.36$, $P = 0.0022$; effect of time since mining on adjusted TBS $F_{1,51} = 11.23$, $P = 0.0015$). However, the rate of increase in TBS is slow, with TBS increasing ~0.01 points/day on average following mining (slope from raw TBS model: 0.012; slope from adjusted TBS model: 0.0135). Recall that post-mining TBS at these sites are, on average, nine points lower than the pre-mining TBS (Figure VII-7). The regression equations from the current analysis indicate that it would take nearly three and half years for a station's TBS to increase by nine points and recover to roughly pre-mining levels (raw TBS model: 1291 days or 3.5 years; adjusted TBS model: 1209 days or 3.3 years). This time to recovery is slightly longer than the three years that the PADEP currently allows for stream mitigation and recovery (Figure VII-6b; PADEP 2005a). For streams experiencing larger reductions in TBS following mining (> 9 points), recovery may take even longer.

Interestingly, the biological recovery does not appear to be a function of recovery in stream chemistry. There was no significant relationship between time since mining and conductivity ($F_{1,35} = 0.04$, $P = 0.8401$) or pH ($F_{1,36} = 1.68$, $P = 0.20$), indicating that water quality does not return to pre-mining levels following mining.

VII.J.3 – Pre- and Post-Restoration Total Biological Scores for Streams with Gate Cut Mitigation

For streams with gate cuts in Bailey and Enlow Fork Mine, the average post-restoration TBS is identical to the average pre-mining score (Figure VII-19). Statistically, there is no significant difference in pre- and post-restoration scores (effect of restoration on raw TBS, $F_{1,58} = 0.01$, $P = 0.94$; effect of restoration on adjusted TBS, $F_{1,58} = 0.32$, $P = 0.57$). These results confirm those of a recent analysis by the mine operator and their consultants, which also found that post-restoration TBS did not differ from pre-mining TBS (Nuttle et al. 2014). Gate cut mitigation appears to be effective in restoring the macroinvertebrate community following mining-induced pooling.

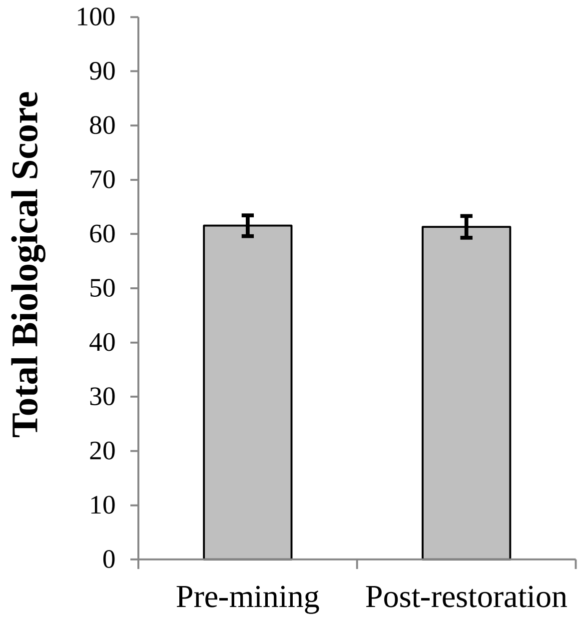

Figure VII-19. Comparison of pre-mining and post-restoration Total Biological Scores for streams with gate cut mitigation in Bailey and Enlow Fork Mines. Data are least squares means +/- 1 standard error.

Similar trends were observed for Cumberland Mine. On average, the raw post-restoration TBS are equal to or greater than the mean control stream score (Figure VII-20). However, it is unclear how indicative these scores are of conditions inside the restoration site due to the distance between the restoration sites and the bio-monitoring stations. It should also be noted that stations DF STA 2 and DF STA 21 have not yet been released from monitoring by PADEP because the post-restoration scores are not within 16% of each other. Post-restoration scores from Bailey and Enlow Fork Mines indicate that two years' worth of monitoring (i.e. roughly four samples) are often required to obtain two samples that score within 16% of each other.

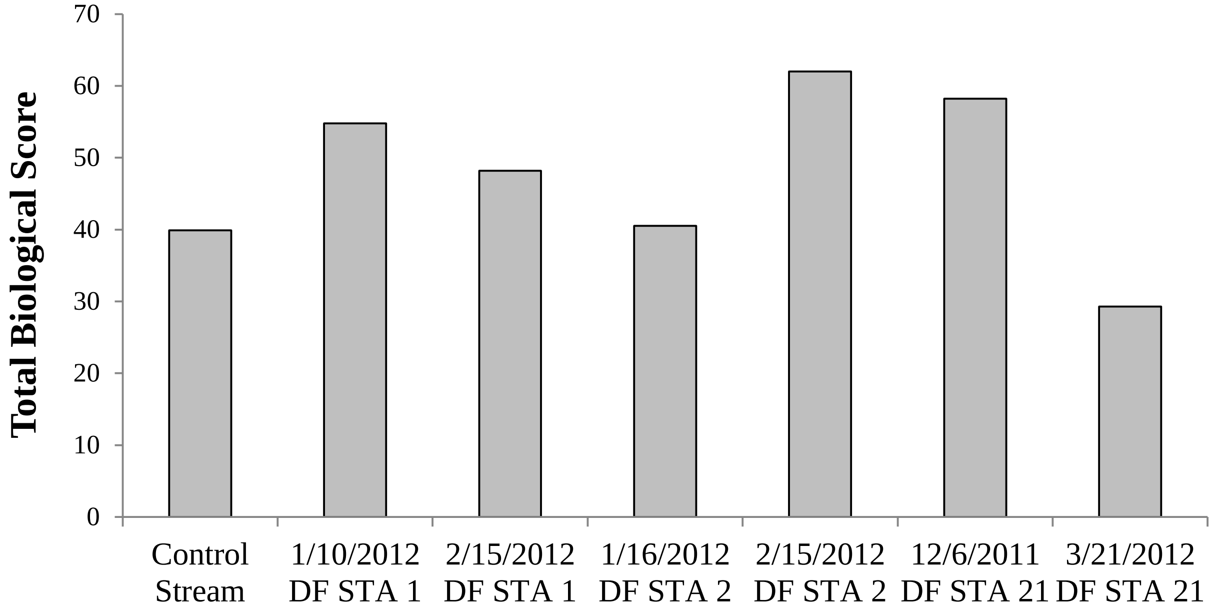

Figure VII-20. Post-restoration Total Biological Scores for three bio-monitoring stations on Dyers Fork in Cumberland Mine. Scores are compared to the mean score from control stream station GAR4. Bio-monitoring stations are ~1,200-ft outside the restoration areas.

VII.J.3 – Verifying TBS Reported by Mine Operators Following Gate Cut Mitigation

To verify the accuracy of the data used in the above analyses and graphs, the University re-sampled four stations where restoration work had been performed and compared the scores to post-restoration scores from the mine operators (Table VII-16). All macroinvertebrate samples were collected in accordance with TGD 563-2000-655 Appendix B methodology. University personnel were trained by a PADEP biologist in TGD sampling methods and certified as being competent (Appendix I1) and a subsample of macroinvertebrate samples were submitted to PADEP for verification of ids (Appendix I2). Data for all University-collected samples can be found in Appendix D2.

Table VII-16. Stations with gate cut restoration that were sampled by the University and compared to samples from the mine operator.

Mine	PA WRDS Stream Code	Stream Name	Station Name	Sampling Date	Within the range of scores reported by mine operator?
Cumberland	41261	Dyers Fork	DF STA 1	3-May-2013	No
Cumberland	41261	Dyers Fork	DF STA 21	3-May-2013	Yes
Enlow Fork	40938	Crafts Creek	BSW20	5-Apr-2013	No
Enlow Fork	32708	Templeton Fork	BSW19	9-May-2013	Yes

Two of the four sites scored within the range of post-restoration scores reported by the mine operator (Table VII-16). The sample for station BSW19 on Templeton Fork scored 41.3, which is similar to post-restoration scores collected by the mine operator in the spring months at this site (44.4 and 41.9; CEC 2012c). The sample for DF STA 21 on Dyers Fork scored 38.6. The mine operator reported scores at this site of 58.2 in December 2011 and 29.3 in March 2012 (Figure VII-20). While the University's score falls within this range, the range is relatively large and reflects a high degree of variability in the scores reported by the consultants.

In contrast, scores for the other two sites fell outside and below the range of scores reported by the mine operator (Table VII-16). The sample from station BSW20 on Crafts Creek scored 34.7. The mine operator has sampled this station three times following restoration, with the following results: 39.2 on 24 May 2011, 51.2 on 1 November 2011, and 60 on 9 April 2012. The University's score is most similar to the first score collected by the mine operator following gate cut mitigation. Differences in the sampling locations may account for the observed differences in TBS. Following restoration, the mine operator re-located station BSW20 slightly downstream so that the station was located directly inside the restoration area (BSW20A; CEC 2012d). The University was unaware of this shift in station location at the time of sampling. Such shifts in sampling location also occurred at Templeton Fork following restoration (BSW19A; CEC 2012c), however, the location change did not affect the score for that site. For DF STA 1, the post-restoration TBS was also lower than the scores reported by the mine operator. Samples collected by the mine operator in January and February 2012 scored 54.8 and 48.2, respectively (Figure VII-20) while the University's sample had a TBS of 32.6. Differences in month of sampling may account for the differences observed at this site.

While a much greater sample size would be needed to draw conclusions regarding the accuracy of data submitted to PADEP as well as the repeatability of TGD 563-2000-655, the University can conclude that sampling location and month of sampling likely play a significant role in determining a station's TBS. The University suggests that PADEP agents continue to monitor sampling efforts by the mine operator and also perform their own spot-checking of the data from time to time, with careful consideration of these factors.

VII.J.4 – Changes in Macroinvertebrate Community Composition for Streams with Gate Cut Mitigation

To determine if gate cut mitigation affects community composition, the University identified nine stations from Bailey and Enlow Fork Mines that reported pre- (N = 14) and post-restoration (N = 21) macroinvertebrate taxa abundance. As in Section VII.H.3, the University focused on taxa in the Ephemeroptera, Plecoptera, and Trichoptera orders and calculated the relative frequency of taxa occurrences among pre- and post-restoration samples. NMDS and permutation tests were also used to determine if restoration significantly affected overall community composition.

While the TBS is nearly identical between pre- and post-restoration samples, there are subtle changes in community composition as a result of restoration. For Ephemeroptera, two moderately common genera (present at more than 50% of sites pre-mining) experienced declines in relative frequency that were >50% (*Ephemerella* and *Ephemera*). Most other Ephemeroptera taxa remained at similar frequencies following restoration (Appendix H2).

For Plecoptera, the University observed a slight shift away from taxa in the families Capniidae and Chloroperlidae but strong increases in genera such as *Perlesta* and *Isoperla* (Appendix H2). It should be noted that genera in the family Chloroperlidae were not common prior to restoration, so the reduction in relative frequency may simply be due to the general rarity of these taxa. However, the genus *Allocapnia* in the family Capniidae was quite common prior to restoration with a relative frequency across sites of 57.1%. Yet, after gate cut mitigation, it was found at less than 25% of sites. *Isoperla*, a widespread genus, showed the opposite trend and nearly doubled following restoration. This genus appears to be highly tolerant of disturbance as it also experienced significant increases following mining-induced flow loss.

Following restoration, the majority of Trichoptera increased in relative frequency across sites (Figure VII-21). In fact, four new genera that were not present prior to mining were identified following gate cut mitigation. Two of these genera, *Ceratopsyche* and *Diplectrona* belong to the family Hydropsychidae.

While there are subtle shifts in EPT taxa occurrences, the NMDS and permutation tests indicate that restoration does not significantly alter community composition (ordination stress = 0.19; time of sampling, P = 0.30; Figure VII-22). Month of sampling was a significant predictor of community composition (R^2 = 0.63, P = 0.001), but station id was not.

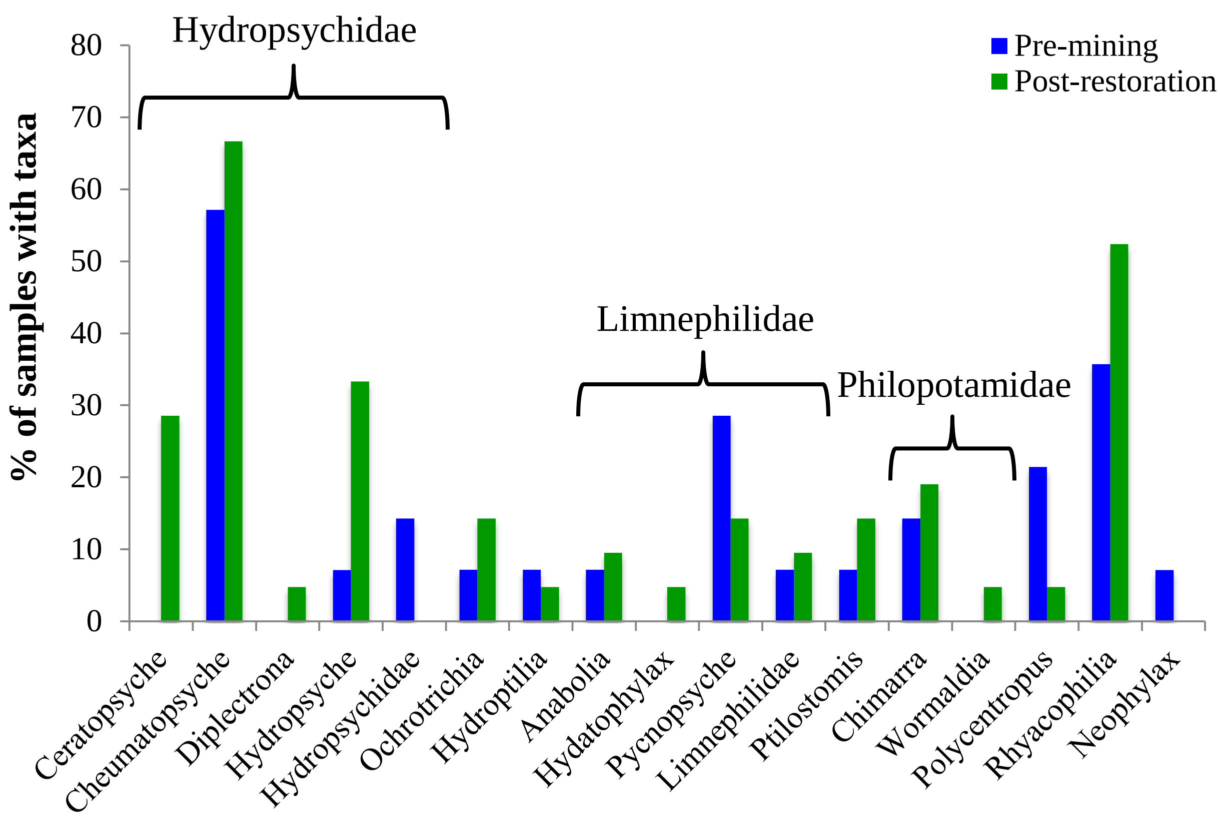

Figure VII-21. Relative frequency (%) of Trichoptera taxa occurrences in pre- (N = 14) and post-restoration (N =21) samples. While the bulk of the samples were identified to the genus level, consultants were at times only able to identify to the family level. For families with more than one genus represented, a bracket is used to group all genera in that family.

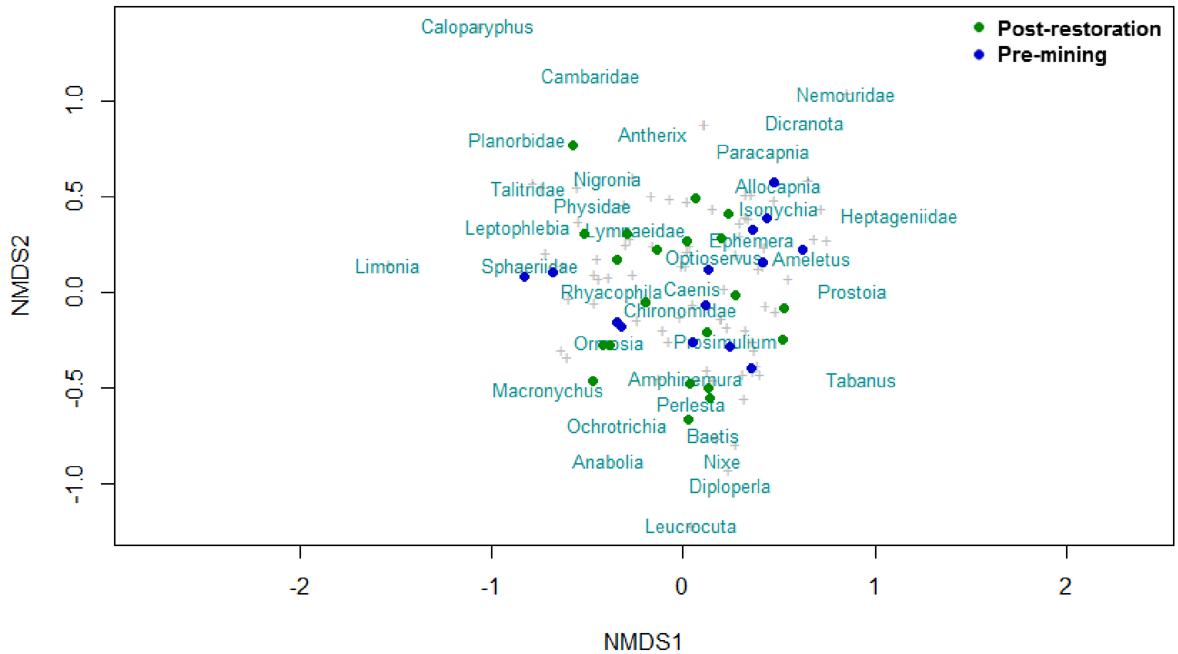

Figure VII-22. Non-metric multi-dimensional scaling ordination for community composition of pre-mining (N = 14) and post-restoration (N = 21) samples for sites with gate cut mitigation. Gray plus signs represent additional taxa whose names are not displayed. When taxa overlap, only the name of the more abundant taxa is displayed.

<u>VII.K – Pre- and Post-Mining Biology in Focal Watersheds</u>

Up to this point, the University's analyses of subsidence impacts on stream biology have utilized data from just one or two mines. Nearly all of the post-mining biology data that was available at PADEP was from Bailey Mine. Similarly, the bulk of the post-restoration data came from just two mines – Bailey and Enlow Fork Mine. To determine if mining-induced flow loss, mining-induced pooling, and restoration work have similar impacts on stream biology at other mines, the University sampled 10 bio-monitoring stations within the Act 54 focal watersheds (Table VII-17). All samples were collected in accordance with TGD 563-2000-655. University personnel were trained and observed by a PADEP biologist in the TGD sampling methodology (Appendix I1) and a subsample of the macroinvertebrate samples were submitted to a PADEP biologist for verification of ids (Appendix I2). Data for all University-collected samples can be found in Appendix D2.

Overall, the University observed a diversity of responses across the sites. Below, a more detailed description of the impacts at each site is provided along with graphs that compare the mine operator's pre-mining scores to the University's post-mining scores.

Table VII-17. Bio-monitoring stations from focal watersheds that were sampled for biology to supplement data from PADEP. Latitudes and longitudes for stations are available in Appendix D2.

Mine	PA WRDS Stream Code	Stream Name	Station Name	Impacted by Mining?
Sites impacted by flow loss and grout mitigation				
Bailey	32547	Strawn Hollow	BSW39	Yes*
Blacksville 2	41813	Roberts Run	BSW22	No
Cumberland	40608	UNT to Maple Run	MRT12	Yes
Enlow Fork	40941	UNT to Crafts Creek	BSW24	Yes
Sites impacted by flow loss but not requiring grout mitigation				
Enlow Fork	40944	UNT to Crafts Creek	BSW13	No*
Sites impacted by pooling				
Blacksville 2	41812	Blockhouse Run	BSW23	Yes
Emerald	41014	Muddy Creek	MC B2	Yes
Sites with gate cut mitigation				
Emerald	41268	Mount Phoebe Run	MP STA 1	Yes
Unaffected sites				
Cumberland	40607	Maple Run	MR4	No
Cumberland	40607	Maple Run	MR5	No

* Augmentation was on at time of sampling or during month prior to sampling

VII.K.1 – Pre- and Post-Mining Biology in Bailey Mine

Because Barneys Run, the focal watershed in Bailey Mine, was already heavily represented in the University's analysis of the effects of mining-induced flow loss, the University opted to sample biology in a similar, but much smaller watershed of Bailey Mine. Strawn Hollow, a tributary to the South Fork of Dunkard Fork, is located over the 12-15I panels of Bailey Mine. Station BSW39 is located in the middle of the Strawn Hollow watershed in the 14I panel, and the drainage area upstream of this station is characterized by dense forest. Based on the land use, field observations, and the pre-mining TBS data, the station appears to have had little human disturbance prior to mining. Following mining, flow loss and bedrock heaving were observed in the 14I panel. Four augmentation points were installed throughout the watershed to maintain flow across the impacted area. The 14I panel was grouted in 2012. The University sampled station BSW39 on 18 April 2013 and found that grout mitigation has not yet been effective in restoring the biological community to pre-mining levels (Figure VII-23). Relative to the pre-mining data, declines were observed in three of the five TBS metrics, including taxa richness, % EPT richness, and intolerant taxa richness (Appendix D2). The TBS for this site was 59.4, which is less than 88% of the mean pre-mining score. These results reinforce the University's findings from Figure VII-7 and further implicate mining-induced flow loss as a significant contributor to adverse effects on stream communities. It should be noted that one augmentation well was on during the University's sampling and was discharging ~10gpm, so this sample does not reflect the natural biological conditions over the 14I panel. The University suggests that future studies follow-up on the flow and biological recovery of Strawn Hollow.

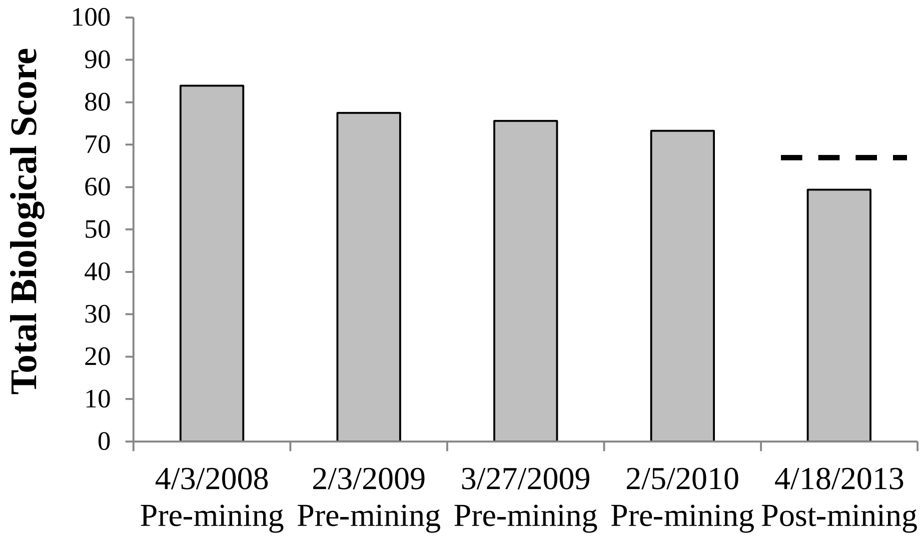

Figure VII-23. Total Biological Scores for bio-monitoring station BSW39 on Strawn Hollow, a tributary to the South Fork of Dunkard Fork, in Bailey Mine. Pre-mining scores were collected by the mine operator while the post-mining score was collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

VII.K.2 – Pre- and Post-Mining Biology in Blacksville 2 Mine Focal Watersheds

The Blockhouse Run watershed runs over the 14-18W panels of Blacksville 2 Mine. It is likely this watershed will continue to be undermined by additional panels in the future due to its relatively large size. Station BSW23 is located in the 15W panel, and while the bulk of the drainage area upstream of this station is forested, there is a small patch of pasture/hay just upstream of the bio-monitoring station. While the pasture/hay has the potential to create human-induced disturbances to the station’s biological community, the one pre-mining score that was available suggests that the community was quite healthy prior to mining (Figure VII-24). Following mining, pooling was observed at this station and several other locations along Blockhouse Run. A gate cut is planned for the 14-15W gate area, which should eventually alleviate the pooling at this station. The gate cut had not been performed at the time of this report. The University sampled this station on 25 April 2013 and found that the pooling at the site was having an adverse effect on the biological community (Figure VII-24). The University’s sample had 12 fewer species than the pre-mining sample and the number of intolerant taxa and filterer-collector/ predators was reduced by 50%. These data add weight to the analysis which suggested that mining-induced pooling impacts adversely affects stream biology communities (Section VII.H.2). However, the decline in TBS observed here is more dramatic than the average declines following pooling that were reported in the analysis. The significant decline in TBS on Blockhouse Run may be a result of the prolonged pooling at this site. Because mining of this station was complete by 26 October 2009, pooling had likely been present on site for > 3 years at the time of the University’s sampling.

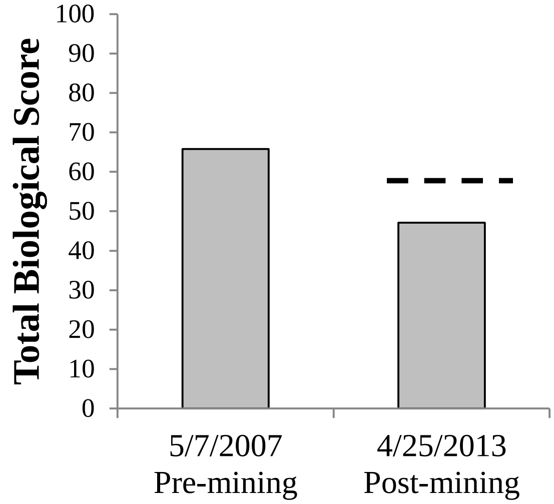

Figure VII-24. Total Biological Scores for bio-monitoring station BSW23 on Blockhouse Run in Blacksville 2 Mine. The pre-mining score was collected by the mine operator while the post-mining score was collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

The Roberts Run watershed is the second focal watershed for Blacksville 2 Mine. During the 4th assessment period, Roberts Run was mined by the 14W panel. Station BSW22 is located roughly 4,000-ft downstream from the headwaters, near the 13-14W gate area. The drainage area upstream of this site is almost entirely forested, with one road running alongside the stream. Again, field observations and the pre-mining data (Figure VII-25) for this site suggest that human-disturbances to the stream biological community are minimal. While no mining-induced changes to the stream channel were noted during mining in fall 2010, a small no-flow section increased in size week by week as the panel undermined the stream. A single augmentation was established to maintain flow over the 14W panel. The stream was grouted in 2012. The University sampled station BSW22 on 25 April 2013 and found that the post-grouting scores were equivalent to the pre-mining scores (Figure VII-25). While these data suggest that grouting may be an effective restoration technique, it should be noted that station BSW22 is ~300-ft downstream from the impacted area. The status of the biological community in the area directly impacted by flow loss is unknown. The University suggests that PADEP continue to monitor this site and while awaiting flow and biology data from the mine operator.

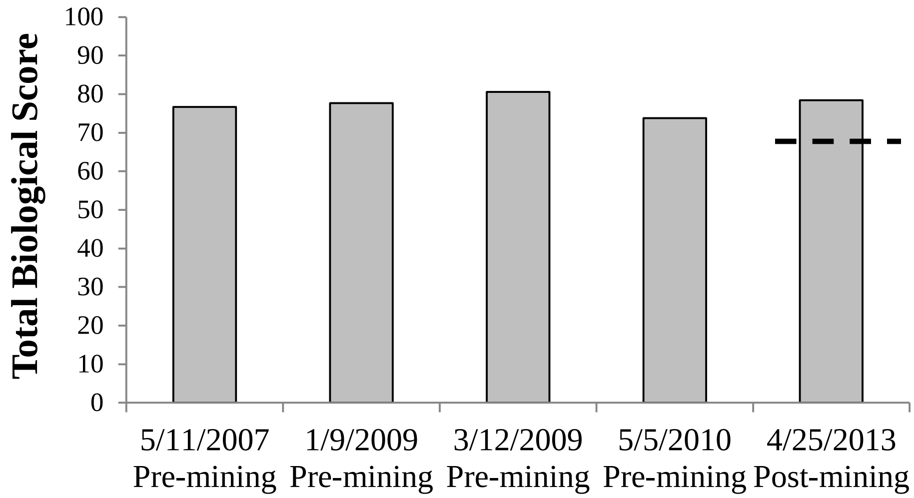

Figure VII-25. Total Biological Scores for bio-monitoring station BSW22 on Roberts Run in Blacksville 2 Mine. Pre-mining scores were collected by the mine operator while the post-mining score was collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

VII.K.3 – Pre- and Post-Mining Biology in Cumberland Mine Focal Watershed

The Maple Run watershed is located in the Cumberland West district over longwall panels 58-61. Within this watershed, the University selected three bio-monitoring stations for sampling – MR4, MR5, and MRT12. MR4 and MR5 are both located along Maple Run, while MRT12 is located on stream 40608, an unnamed tributary to Maple Run. The NLCD classifies the land surrounding Maple Run and stream 40608 as forest (Fry et al. 2011), however during field observations, the University noted that stations MR4 and MR5 were surrounded by pasture/hay, with a residence also very close to the stream. While all streams within this watershed are classified as high quality-warm water fisheries (Table I-2), it is possible that human impacts may explain the moderate to low pre-mining TBS at MR4 and MR5 (Figure VII-26). Following mining, PADEP agents noted that there were no mining-induced changes to Maple Run. Indeed, when the University sampled MR4 and MR5 on 30 April 2013, the post-mining samples were well within the range of pre-mining scores for the two sites (Figure VII-26). It should be noted that the two pre-mining scores for MR4 are not within 16% of each other as required by TGD 563-2000-655. However, these data suggest that macroinvertebrate communities are largely undisturbed at sites where no mining-induced changes are observed.

Station MRT12 is surrounded by forest and the pre-mining scores for this station suggest that the site experiences little human disturbance and had a healthy macroinvertebrate community prior to mining (Figure VII-27). Following mining, bedrock fractures and compression heaves were noted in stream 40608, just downstream of station MRT12. These mining-induced changes resulted in a flow loss impact. Bentonite clay was used to fill the fractures and mitigate the damages. To the University’s knowledge, augmentation was not used at this site. The University sampled MRT12 on 8 March 2013 and the sample indicates that the site is experiencing an adverse effect from mining. The sample score is less than 88% of the pre-mining average (Figure VII-27). While pre-mining samples always identified ≥ 25 taxa at this site, the University’s sample only identified 19 unique taxa. Data from this site confirm the analysis from Figure VII-7 and indicate that mining-induced flow loss generally reduces stream TBS.

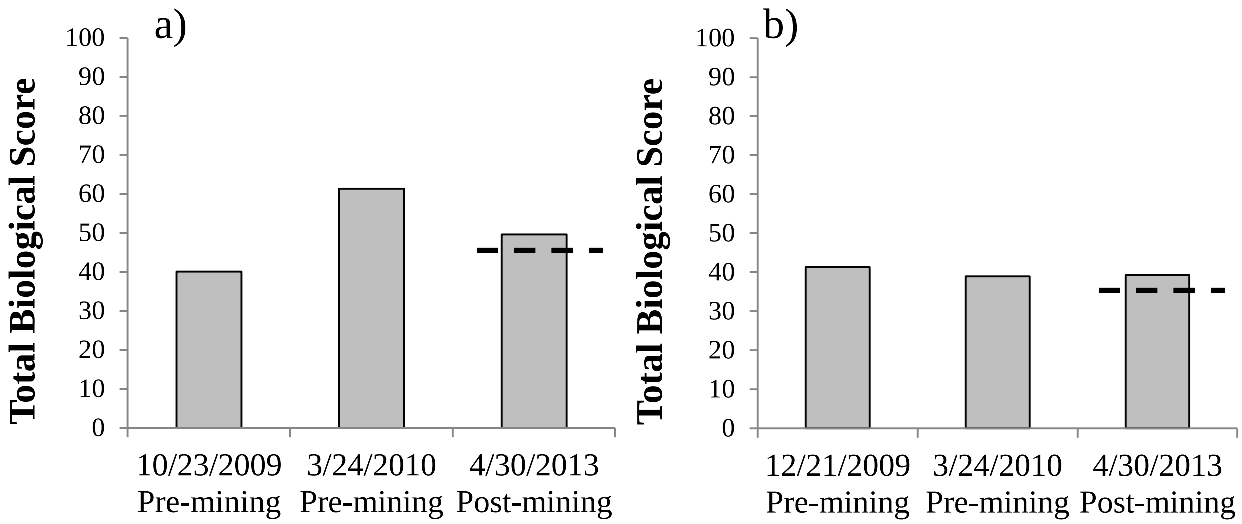

Figure VII-26. Total Biological Scores for a) bio-monitoring station MR4 and b) station MR5 on Maple Run in Cumberland Mine. Pre-mining scores were collected by the mine operator while post-mining scores were collected by the University. Dashed lines indicate the minimum Total Biological Score required for stream to be considered recovered.

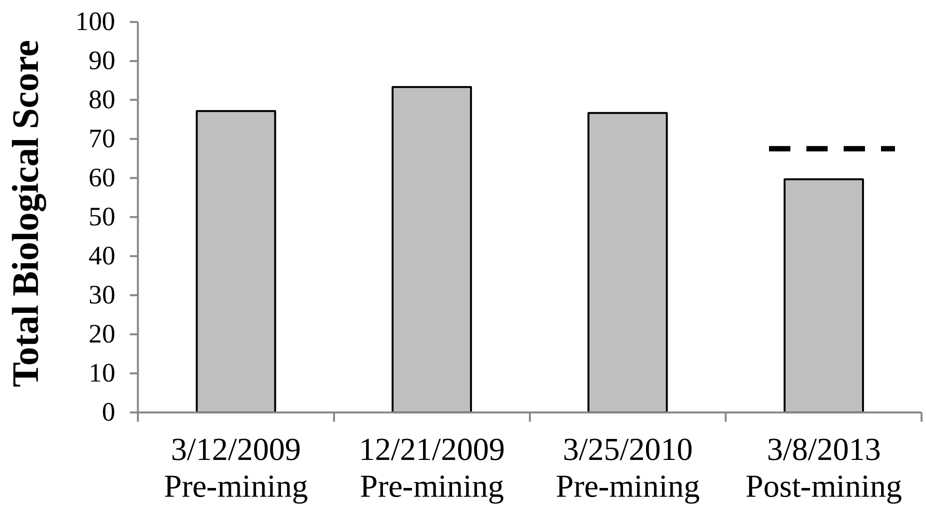

Figure VII-27. Total Biological Scores for bio-monitoring station MRT12 on stream 40608, an unnamed tributary to Maple Run, in Cumberland Mine. Pre-mining scores were collected by the mine operator while the post-mining score was collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

VII.K.4 – Pre- and Post-Mining Biology in Emerald Mine

The University did not select any focal watersheds in Emerald Mine, however, it is important to sample sites across all longwall mines in an equal manner. During the 4th assessment period, the

Mount Phoebe Run watershed was undermined by panels B6 and B7 in Emerald Mine. Bio-monitoring station MP STA 1 is located ~1,200-ft south of the B7 panel on Mount Phoebe Run. Field observations revealed that at this site the stream is bounded on the left side by forest and on the right side by a residence. The banks of the stream near the residence are lined with tall herbaceous cover although some large pieces of equipment are also piled nearby. Further upstream, the drainage area becomes characterized by a mix of forest and pasture/hay. The direct adjacency of station MP STA 1 to a residence suggests that there is likely some degree of human disturbance on the stream site. Unfortunately, pre-mining data are not available to confirm this. The pre-mining scores for MP STA 1 were not within 16% of each other and thus were not provided by the mine operator to PADEP. Following mining of the B7 panel in March 2010, pooling in excess of 1-ft developed near the southern end of the B7 panel. The pooling had not been predicted by subsidence modeling, so a permit revision was required for the gate cut mitigation project. The gate cut was completed in November 2012. The University sampled station MP STA 1 on 8 November 2013 to evaluate the recovery of the biological community at this site. While pre-mining scores were not available, the mine operator proposed using pre-mining scores from MP 3, a nearby station upstream of the gate cut mitigation area, as a restoration target for MP STA 1. The University's sample shows that the post-restoration TBS from MP STA 1 is much lower than the pre-mining scores at MP 3 (Figure VII-28). This difference may reflect an unrecovered macroinvertebrate community, differences in month of sampling, or a poor choice of "restoration target" scores. However, it should be noted that as with the bio-monitoring stations along Dyers Fork, MP STA 1 is located some distance away from the restoration area. It is unclear how well conditions at MP STA 1 reflect biological recovery in the mitigated area. The University strongly suggests that the mine operator establish bio-monitoring stations within future gate cut mitigation sites.

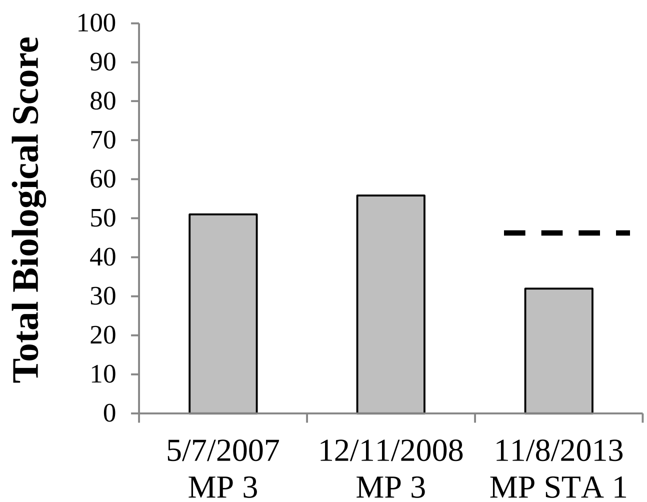

Figure VII-28. Total Biological Scores for bio-monitoring stations MP 3 and MP STA 1 on Mount Phoebe Run in Emerald Mine. Pre-mining scores for MP 3 were collected by the mine operator while post-restoration scores for MP STA 1 were collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

The Muddy Creek watershed was undermined by panels C2-C3 of Emerald Mine during the 4th assessment period. Station MC B2 is located on Muddy Creek in the C2 panel, near the mouth of stream 41085 (an unnamed tributary to Muddy Creek). The drainage area above MC B2 contains a mix of forest, pasture/hay and row crops. The area directly adjacent to the station is largely pasture/hay. The moderate pre-mining scores at this site suggest that land use practices were impacting the stream community before mining began (Figure VII-29). Following mining of the C2 panel, pooling was observed by PADEP agents near station MC B2. Mitigation measures have not yet been taken to alleviate the pooling. The University sampled station MC B2 on 15 November 2013 and the sample scored a TBS of just 31.1. This score is not within 88% of the pre-mining scores, and it is much lower than post-mining scores collected by the mine operator. The differences between the University's post-mining score and those of the mine operator may be due to differences in month of sampling. The University's score was collected in November, a month during which samples score on average 10-11 points lower than samples collected during spring months (Figure VII-4).

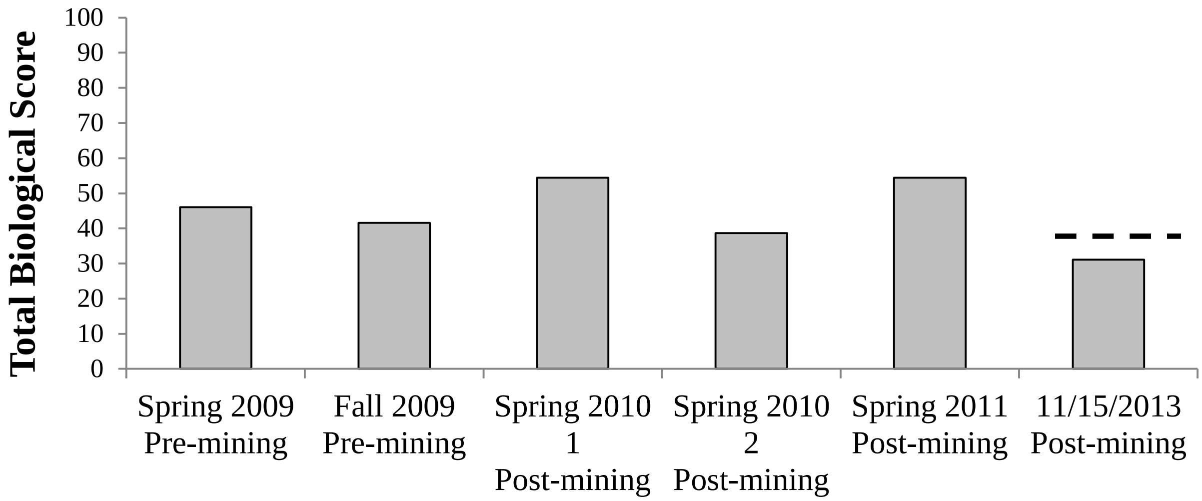

Figure VII-29. Total Biological Scores for bio-monitoring station MC B2 on Muddy Creek in Emerald Mine. Pre and post-mining scores for MC B2 were collected by the mine operator and a single post-mining score was collected by the University in November 2013. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

VII.K.5 – Pre- and Post-Mining Biology in Enlow Fork Mine Focal Watershed

The Crafts Creek watershed in Enlow Fork Mine was undermined by longwall panels E15 through E23 during the 3rd and 4th Act 54 assessment periods. The University selected two bio-monitoring stations for sampling within this watershed. Station BSW24 is on stream 40941, a relatively short first order tributary over panels E20 and E21. Station BSW13 is on stream 40944, another first order tributary that crosses panels E16-E19. Station BSW13 is located in the E18 panel. The Crafts Creek watershed contains a mosaic of three major land use types – forest, pasture/hay, and row crops. While the drainage basins for these stations are largely forested, the area upstream of station BSW13 on stream 40944 contains ~20% pasture/hay land use. These land use practices, along with the many residences and recreational activities in the watershed, suggest that the Crafts Creek watershed experiences more human influences than any of the other focal watersheds. The moderate pre-mining scores (Figure VII-30, Figure VII-31) also indicate that this watershed had some degree of human disturbance prior to mining.

Stream 40941 experienced mining-induced flow loss impacts in January 2010 after mining of the E20 panel. An augmentation discharge point was established at the E20/E21 gate to maintain flow and the affected area was grouted in November 2010. The University sampled station BSW24 on 5 April 2013 and found that this station was adversely affected by flow loss. The post-mining TBS of 47.2 is much lower than the pre-mining score of 64.9 (Figure VII-30). Interestingly, the sample was dominated by amphipods of the genus *Crangonyx* (112 *Crangonyx* individuals out of 170 total individuals; Appendix D2). This finding is somewhat unusual and further indicates that the community structure at this site has been significantly altered by mining, as amphipods were not present in the pre-mining sample. Because augmentation has not been used at this site since fall 2011 (based on augmentation submitted by the mine operator), the University is confident that this sample represents the natural biological conditions on stream 40941.

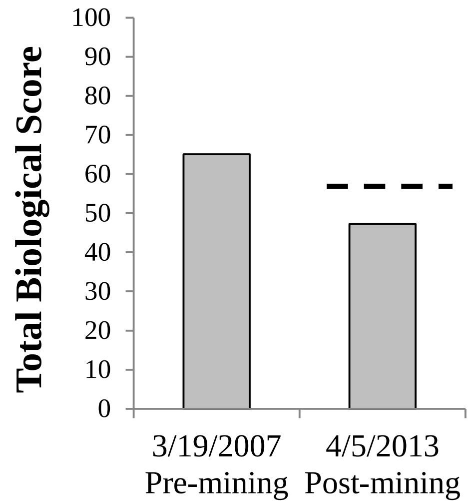

Figure VII-30. Total Biological Scores for bio-monitoring station BSW24 on stream 40941, an unnamed tributary to Crafts Creek. The pre-mining score was collected by the mine operator while the post-mining score was collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

Stream 40944 and bio-monitoring station BSW13 were undermined by panel E18 in January-February 2009. While some flow loss issues have been noted in the stream section above this panel, the most significant flow losses on stream 40944 occurred downstream over panel E17. Augmentation in the E18 panel has been instrumental in maintaining flow through the heavily impacted E17 panel area. The section over the E18 panel itself however has never received grouting or any mitigation other than augmentation. The University sampled station BSW13 on 29 March 2013 and the post-mining TBS fell well within the range of the pre-mining scores collected at this site (Figure VII-31). While this is encouraging, it should be noted that augmentation was used extensively on this stream in fall 2012 through February 2013. While augmentation was not on during the University’s sampling, the sample may not accurately reflect the natural biological conditions on site.

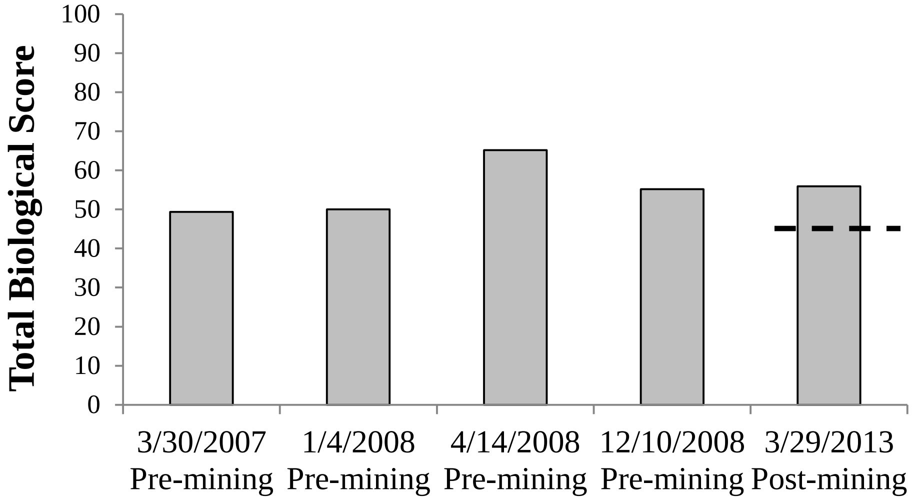

Figure VII-31. Total Biological Scores for bio-monitoring station BSW13 on stream 40944, an unnamed tributary to Crafts Creek, in Enlow Fork Mine. Pre-mining scores were collected by the mine operator while the post-mining score was collected by the University. Dashed line indicates the minimum Total Biological Score required for stream to be considered recovered.

VII.L – Summary

To isolate the effect of mining on stream flow and biology, it is critical to account for factors that cause natural variation in stream ecosystems. Stream flow can be influenced by climate, so in this assessment, stream flow losses are reported for both the wet (December – May) and dry (June-November) seasons. Stream biology can be influenced by watershed and reach-level characteristics as well as month of sampling. Low TBS are generally associated with watersheds with pasture/hay land use and reaches with poor habitat scores and alkaline pH. TBS are also lower when stream samples are collected in early fall (i.e. October and November). In this assessment, variation in watershed and reach characteristics as well as month of sampling are controlled for when analyzing the impacts of mining on stream macroinvertebrate communities.

During the 4th Act 54 assessment, 96.05 miles of stream were undermined, which represents a 16% decline from the 3rd assessment. Longwall mining accounted for the undermining of 50.59 miles of stream undermining. Of the stream miles undermined by longwall techniques, 39.2 miles, 77%, belong to streams that experienced mining-induced flow loss, pooling, or both somewhere along their channel. On these streams, the maximum length of post-mining flow loss ranged from 936-ft to 10,883-ft in the dry season and from 96-ft to 8,106-ft in the wet season. The length of mining-induced pooling on individual streams could not be estimated due to a lack of data.

PADEP no longer has a stream investigation period in which they determine if changes in flow are related to mining or climate. Instead, changes in flow that occur at the time of mining are automatically assumed to be mining-related.

PADEP is responsible for tracking all mining-induced impacts on stream flow and biology. PADEP's methodology for tracking stream impacts changed between the 3rd and 4th Act 54 assessments as a result of the implementation of TGD 563-2000-655. Currently, stream investigations are only used by PADEP to track impacts that occurred before TGD 563-2000-655. For impacts occurring after this point, PADEP no longer requires a formal investigation to determine if the changes in flow are climate or mining-related - instead, the mine operator is automatically assumed liable for impacts occurring at the time of undermining. Once an impact occurs, a record is made in BUMIS and in the SSA stream data logs and the operator is given three years to mitigate the impact and submit data to PADEP for review. The data submission initiates a stream recovery report at PADEP. If the data in the stream recovery report indicates recovery, then the stream is released. If the stream is not recovered, then PADEP can request a change in future mining plans. At this time, the mine operator has two more years to perform additional mitigation work before PADEP will require compensatory mitigation. Thus five years of mining can continue under the existing permit before a final determination of recovery or lack thereof is made. In the University's assessment, this time period may prevent the PADEP from taking action to prevent permanent stream flow loss on additional streams when mining conditions, overburden depth and composition and other factors are similar to those leading to unrecoverable stream loss in the first instance.

As a result of these changes, the 4th assessment period saw just nine stream investigations by PADEP. Of these, four were unresolved at the end of the assessment period. For many of the stream investigations, flow data from the mine operator was inadequate to assess recovery. Fourteen stream recovery reports were filed during the 4th assessment. PADEP has released nine of these from further monitoring, while two cases require compensatory mitigation, and three cases are still under review. In the resolved cases, the University noted that PADEP would occasionally use one post-mining TBS rather than two to assess biological recovery on a stream.

PADEP tasked the University with assessing pre- and post-mining Total Biological Scores (TBS) for at least five stream segments that experienced mining-induced flow loss and at least five stream segments that experienced mining-induced pooling. For the flow loss investigation, Bailey Mine was the only mine with both pre- and post-mining biology data that met the requirements of TGD 563-2000-655. In this mine, mining-induced flow loss significantly reduces TBS and the decrease constitutes an adverse effect under the definition given in TGD 563-2000-655. Mining-induced flow loss drives declines in Ephemeroptera and taxa in the families Ephemerellidae and Heptageniidae appear to be especially sensitive. Post-mining macroinvertebrate communities are characterized by a shift to shredder and predator taxa and stress-tolerant Dipteran taxa. The changes in biology are accompanied by changes in water quality, with conductivity and pH significantly increasing at sites with mining-induced flow loss. On average, the increases in conductivity exceed the U.S. EPA's benchmark for aquatic life in the Western Allegheny Plateau ecoregion. For the pooling investigation, data from Bailey and Enlow Fork Mines indicate that mining-induced pooling reduces TBS. When adjusting TBS for the effects of watershed and reach-level characteristics and month of sampling, pooling impacts also constituted an adverse effect to streams.

As the first Act 54 report to quantitatively track stream mitigation, the University found that:

- 28 stream segments received gate cuts to alleviate mining-induced pooling (Total miles mitigated: 4.21 miles).
- 95 streams had augmentation discharges installed along their channel and augmentation was active at 74 of these streams to maintain flow during or after mining.
- 57 streams received grouting to mitigate mining-induced flow loss. Estimates for one longwall mine suggest that ~50% of the stream miles undermined received grouting.
- Three stream segments had liners placed in their channels to restore flow following mining-induced flow loss.
- At one longwall mine, ~7,913-ft of access roads were constructed immediately adjacent to streams in just a three month period to support mitigation activities.

In assessing the effectiveness of stream mitigation techniques, the University found that TBS increases over time at sites experiencing mining-induced flow loss. The rate of recovery is slow though, and analyses suggest that on average, 3 ½ years are required to restore the macroinvertebrate community to its pre-mining condition. In contrast, water quality does not recover over time and pH and conductivity at flow loss sites remain elevated following mitigation. For streams with pooling impacts and gate cut mitigation, restoration is effective in restoring TBS to pre-mining levels. Macroinvertebrate community composition following gate cuts is indistinguishable from the pre-mining community composition.

Data from the focal watershed analyses generally supports the conclusions above, but also highlight the site-specific nature of mining impacts. Stream characteristics and mitigation measures differ from site to site. In general, for sites that were unaffected by mining, post-mining TBS matched pre-mining TBS.

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