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To cite this article: Matthew R V Ross *et al* 2021 *Environ. Res. Lett.* **16** 075004

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ENVIRONMENTAL RESEARCH
LETTERS

LETTER

OPEN ACCESS

RECEIVED

11 December 2020

REVISED

1 June 2021

ACCEPTED FOR PUBLICATION

9 June 2021

PUBLISHED

29 June 2021

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Mountaintop mining legacies constrain ecological, hydrological and biogeochemical recovery trajectories

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E-mail: mrvr@rams.colostate.edu**Keywords:** mountaintop mining, recovery, legacy, weathering, hydrology, remote sensing**Abstract**

Mountaintop mining, like all forms of surface mining, fundamentally alters the landscape to extract resources that lie 10–100 m below the land surface. Despite these deep, critical zone alterations, post-mining landscapes are required by United States law to be restored to ecosystems of equal or greater value than the ones they replace. Yet, remote sensing of vegetation across more than 1000 km² of reclaimed surface mines in WV, USA reveals little evidence that these habitats are returning to the diverse Appalachian forests that were removed by mining. Instead, even decades after reclamation, mined landscapes are dominated by shorter and sparser trees. Based on detailed field studies and literature synthesis, we suggest that part of these widespread failures in re-establishing native forest result from the fundamental changes in critical zone processes on the post-mining landscape. Former surface mines have substantially altered topography, hydrology and chemistry. In these post-mining, synthetic landscapes, water moves more slowly through piles of exploded bedrock, changing the system from one dominated by stormflow in unmined catchments, to one dominated by baseflow after mining. This slow-moving water, travelling through high surface-area debris and pyrite-rich bedrock, creates ideal conditions for highly elevated weathering in mines both old and new. These foundational changes to the critical zone set ecosystem recovery along a novel trajectory, in which the legacy of past disturbance is likely to constrain the establishment of native forest for many decades.

1. Introduction

More than 400 000 km² of the Earth's surface have been mined to extract coal, metals, and rare earth elements (Hooke *et al* 2012). This process requires altering the topography, surficial geology, and soils in the landscape to reach ores. Surface mines penetrate so deeply below the land surface, that their impacts on ecosystems are more akin to volcanic eruptions or meteor impacts than they are to more common disturbances such as timber harvest or burning (Ross

et al 2016). Yet, efforts to reclaim and revegetate mined landscapes have been strongly guided by theories of ecological succession developed in these less extreme forms of land cover change (Oosting 1942).

In the USA, these theories of ecological succession have become codified in the Surface Mining Control and Reclamation act of 1977 (SMCRA) which requires restoring mined landscapes to their prior vegetative state and approximate original contour (US Congress 1977). The law states that mine operations must:

‘establish on the regraded areas, and all other lands affected, a diverse, effective, and permanent vegetative cover of the same seasonal variety native to the area of land to be affected and capable of self-regeneration and plant succession at least equal in extent of cover to the natural vegetation of the area; except, that introduced species may be used in the revegetation process where desirable and necessary to achieve the approved postmining land use plan’ (from section 515b (19)).

Essentially, SMCRA requires coal mine operators to undertake large-scale restoration in order to return the mined landscape to its former state. It is clear from the language of the act, that ecological succession models (Cowles 1911, Clements 1916, Wright and Fridley 2010) are central to these recommendations, and that lawmakers assume that vegetation succession, assisted by initial regrading and revegetation, will lead to ‘natural vegetation’ cover.

Despite these regulations and significant research investments, native forest recovery efforts consistently fail on the surface coal mines of Central Appalachia (Rodrigue *et al* 2002, EPA 2005, Zipper *et al* 2011, Skousen and Zipper 2021). In this region, mountaintop mining (MTM) for coal extraction has become the dominant driver of land-use change, with more than 6000 km² of land converted to active and completed surface coal mines over 30+ years (Pericak *et al* 2018). Throughout the history of MTM, there have been efforts to understand the ecological parameters that might prevent native forest recovery. These studies have suggested that the hydroseeding of nonnative grasses, used to stabilize soils, prevents subsequent growth of native trees through competitive exclusion (Schoenholtz *et al* 1987, Torbert *et al* 1995, Skousen *et al* 2009, Zipper *et al* 2011, Franklin *et al* 2012), and that high rates of herbivory by deer can be a primary constraint to tree growth (Skousen *et al* 2009, Lituma *et al* 2021).

While ecological interaction such as the planting of non-native foundation species or the presence of deer may prevent successful tree growth, we suggest that the massive physical alteration of mined landscapes is another potential significant constraint to forest regrowth (Nippgen *et al* 2017, Ross *et al* 2018, Brooks *et al* 2019, Naslund *et al* 2020, Jaeger and Ross 2021). Mine reclamation efforts attempt to grow trees on the surface of valley fills, collections of unconsolidated rock that can be 100 m deep (Ross *et al* 2016) or on constructed plateaus that often have highly compacted surface soils (Evans *et al* 2015, Greer *et al* 2017), in a landscape with fundamentally novel shapes and geomorphology (Jaeger and Ross 2021). Below the surface, unconsolidated spoil

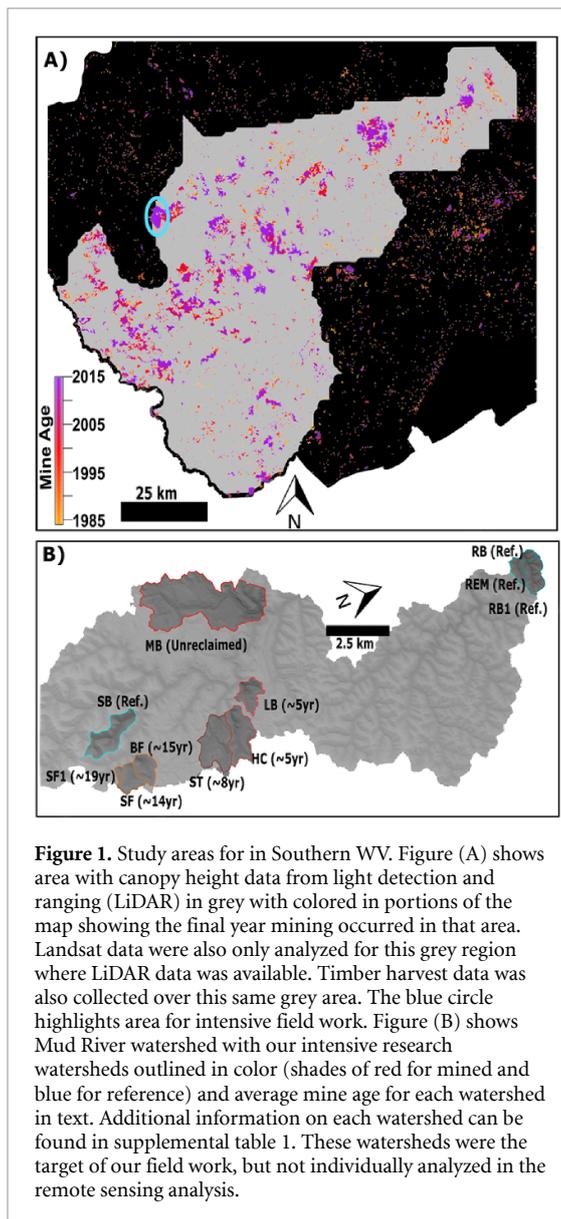
material allows for the development of deep and slow flow paths for water (Nippgen *et al* 2017), while coal and shale residues within the fill generate sulfuric acid that weathers 20–50× faster than in unmined landscapes (Ross *et al* 2018). Together, the altered hydrology and enhanced chemical weathering generate highly saline and alkaline groundwater that is likely below the rooting zone.

Rather than assuming that mine reclamation and reforestation can be guided by models of secondary succession, we suggest that restoring mined lands, like other critical zone disturbances, must instead be guided by theories of ecosystem development and soil formation (Odum 1969, Jenny 1994). Hans Jenny identified five critical soil forming factors: climate, organisms, topography, parent material, and time (Jenny 1994), each of which are altered by surface mining. Earth-moving processes change the topography by lowering average slopes, altering surface and subsurface flowpaths, and creating entirely novel landforms (Maxwell *et al* 2012, Ross *et al* 2016, Reed and Kite 2020). By exploding, transporting and reconsolidating parent material, surface mines both increase the surface area exposed to the elements and redistribute and recombine the minerals found on both the surface and subsurface. The dramatic alterations of topography and porosity in mined landscapes can lead to significantly longer median water residence times as water moves slowly through exploded bedrock at depth (Zegre *et al* 2014, Nippgen *et al* 2017). To our knowledge there are no studies that take a unified critical zone approach (Richter and Billings 2015) to understanding how the deep alterations from MTM will constrain vegetation and ecosystem recovery trajectories. Our research is the first to unify the factors that impact ecosystem recovery by asking the question:

what are the linked ecological, hydrological, and biogeochemical recovery trajectories in mined landscape?

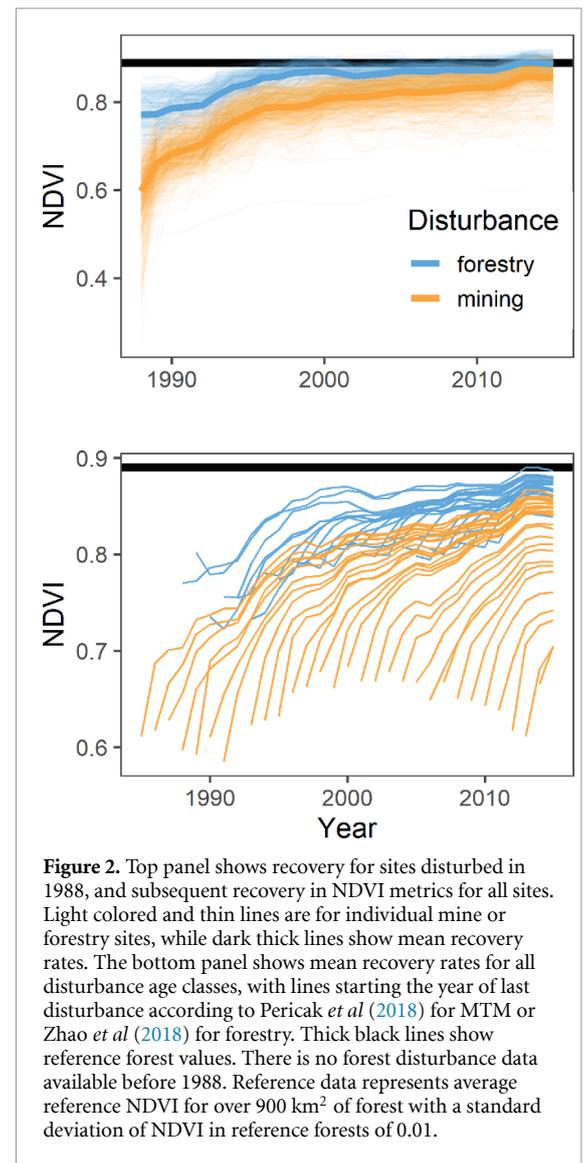
2. Methods

We used two separate, complementary analyses across different spatial and temporal scales to answer our research question. We first conducted a regional analysis of forest recovery (>11 500 km²; figure 1) over multiple decades. We then carried out a separate in-depth analysis at the watershed scale (<10 km²; figure 1(B)) over one water year (1 October 2015–2016). At the regional scale, we used remote sensing tools coupled with information on mine age (Pericak *et al* 2018) to measure changes in vegetation cover and stature over time (figure 1(A)). We conducted our field study of the hydrologic and biogeochemical impacts of MTM in the Mud River watershed. The Mud River is a tributary to the Guyandotte River and is located in Southwestern WV, USA (figure 1). Seven



mined watersheds were selected to span a wide range of time since mine abandonment and differences in valley fill volume (500 000–127 000 000 m³) and were compared with two reference watersheds of similar size (figure 1(B)). Hydrologic and biogeochemical exports from the seven mined watersheds were compared to two mixed deciduous hardwood forest reference watersheds, the 118 ha Rich's Branch (RB) and the 135 ha Spring Branch. The reference watersheds have shallow (<2 m) soils (Sucre *et al* 2011), and are underlain by geology of layered sandstone, siltstone, and shale (Dicken *et al* 2005).

The seven mined watersheds range in current size from 2.5 ha to 590 ha (figure 1(B)), with some watersheds having doubled or halved in size as a result of ridge removal during mining activities. Valley fill volume in these watersheds vary from 0.5 to 16 million m³ (data from Ross *et al* 2016). We calculated mine age for each watershed by averaging the last-mining year for every 30 m × 30 m grid cell within



each watershed outline. This approach yields mine ages that range from unreclaimed 0 year to 19 years old watersheds. Thus, we capture a gradient of watershed size, mine age, and topographic and geologic change (figure 2). Vegetation on these watersheds was a mix of grassland and bare earth cover on the younger mine sites, and a nascent forest of Autumn and Russian olive and Black Locust on our older mine sites.

2.1. Vegetation analysis

We used two complementary approaches to assess vegetation recovery post-mining: satellite remote sensing of greenness, and vegetation height and canopy structure using LiDAR data. Using a dataset derived from Pericak *et al* (2018), mined polygons were split based on the year in which they were last detected as being actively mined. We used Google Earth Engine to create annual, leaf-on (April–September) median composites of Landsat Surface Reflectance data, from 1984–2019. The Landsat composites were harmonized using the reduced major axis

regression coefficients as defined by Roy *et al* (2016) for surface reflectance products. For each annual composite, we calculated a greenness index, the normalized difference vegetation index (NDVI) using the equation:

$$\text{NDVI} = \frac{\text{NIR} - \text{Red}}{\text{NIR} + \text{Red}}$$

NDVI is primarily a measure of the percentage cover of green leaves on a landscape. We directly compared mined land recovery to both reference forests and forests that were disturbed but not in our mining dataset or within mining permit boundaries (data from Zhao *et al* 2018).

In conjunction with the 30 m resolution mine age dataset, we developed a canopy height dataset for the portion of WV with available comprehensive LiDAR data generated in 2010 (data details in Ross *et al* 2016), data available in area shown in grey in figure 1. LiDAR is a highly accurate way to estimate both surface elevation and vegetation height (Hurtt *et al* 2004). Using the raw LiDAR dataset, we used the open-access Fusion software (McGaughey 2012) to generate a 5 m resolution vegetation height map, where vegetation height was calculated as the 95% height maximum of the LiDAR point cloud to conservatively estimate maximum canopy height over a 5 m resolution pixel. The 5 m resolution coverage was aggregated to the 30 m resolution of mine age by taking the median canopy height of all 36 cells within a 30 m pixel. For an estimate of grassland/shrub cover, we also aggregated the 5 m resolution data by calculating the total amount of land with a max canopy height of less than 2 m in each aggregated 30 m cell. This vegetation height data was then combined with the mine age dataset to generate a space-for-time substitution study to explore canopy height changes over time. Because valley fills have unique soil and vegetation makeup compared to non-valley fills (Wunsch *et al* 1999, Greer *et al* 2017), we also grouped the response by landscape type (valley fill or not valley fill). We analyzed this LiDAR data using a space-for-time substitution, where mined areas were grouped by the year of last mining activity, and analyzed as a function of time since last mining. As such, all LiDAR analysis was only performed on sites that had been mined before 2010 (when the LiDAR was flown).

2.2. Hydrologic response

The study period covers the 2016 water year, (1 October 2015–2016). Precipitation was measured at five different locations (figure 1(B)) using Onset HOBO rain gauges recording at 10 min intervals and watershed areal precipitation was derived via inverse distance weighting. Any missing rain or water level data (<5% for WY 2016) was filled using a double mass curve approach with adjacent rain gauges. Water level and stream specific conductance (SC) in each

watershed were measured with Onset HOBO water level loggers and SC. At RB, we also instrumented four sub-watersheds that ranged in size from 2 ha to 70 ha (figure 1(B)). We generated rating curves with stage-discharge relationships via field-velocity profiling and salt dilution in the open channels for five watersheds (RB (ref), MB (0 year), LB (5 years), BF (15 years), and SF (15 years)) across the full range of stage measurements, with a minimum of eight manual discharge measurements per site. We were unable to measure discharge at the highest water levels for the three remaining sites (MB, BF, and SF) and instead used bank-full Gauckler–Manning estimates to populate the rating curves at high water levels. At these sites, discharge was lower than bankful Q 95% of the time. We used the same methods as previous work at these sites to partition discharge between baseflow and stormflow (Nippgen *et al* 2017).

In addition to analyzing sites with discharge data, we also used a combination of SC (a measure of salinity in the stream) sensor data and level data to estimate the percent of zero-flow time at a larger set of watersheds and sub-watersheds where we had installed sensors ($n = 16$). At these sites, we assigned a zero-flow estimate if SC data was reading zero (indicating the absence of water for the sensor to measure SC) and water level either read zero or, if the sensor was installed in a pool, the recorded value was below levels that were confirmed in the field as zero active in-stream flow.

2.3. Biogeochemical response

We previously documented extremely high weathering derived solute fluxes from one mined watershed (LB in this paper) as compared to very low ion fluxes in a similarly situated reference watershed (RB in this paper, Ross *et al* 2018). Here, we add to that assessment by comparing ionic strength and solute fluxes across seven mined and two reference watershed streams at a ten-minute interval for a full year. Sensors were cleaned monthly and checked against calibrated SC measurements. Sensor drift between cleanings, caused by ion deposition, was assumed to be linear and corrected accordingly.

In previous work, we have shown that individual ion concentration can be modeled using the relationship between SC and baseflow Q (Ross *et al* 2018). We collected biweekly water samples to build Q to SC relationships at our five primary watersheds. These repeat sampling efforts were complimented by sampling campaigns during storms in October, April, and June of the 2014–2015 water year (Ross *et al* 2018). Samples were field filtered and stored at 4 °C before analysis of anion and cation concentrations on two Dionex ICS-2000 ion chromatographs with an AS-40 autosampler (Dionex, Sunnyvale, CA). Anions were analyzed on AS-18 guard and analytical columns. Minimum detection was 10 ppb for Cl^- and SO_4^{2-} and 3 ppb for NO_3^- -N. Cations were analyzed

on CS-12 A guard and analytical columns. Minimum detection was 0.3 ppm for Ca^{2+} , Mg^{2+} and Na^+ and 30 ppb for K^+ . All samples with SO_4^{2-} concentrations greater than 200 ppm were diluted prior to analysis. We then converted concentrations to moles and charge equivalents to calculate HCO_3^- concentrations. We assumed that all unmet negative charge could be attributed to HCO_3^- (Ross *et al* 2016) such that in eq/L:

$$[\text{HCO}_3^-] = \Sigma \text{cations} - \Sigma \text{anions}$$

All statistical analyses and figures were generated using R statistical software (R Core Team 2013). Other key packages used included: raster for terrain and vegetation height analyses and tidyverse family of packages for data wrangling, plotting, and organization (Wickham *et al* 2019).

3. Results

3.1. Vegetation recovery

A critical aspect of SMCRA that governs mine reclamation is the need for vegetation to be returned to 'at least equal extent of cover' to natural vegetation. We assessed this structural recovery of mined lands using NDVI and canopy height, both metrics that directly relate to vegetation cover. These metrics clearly demonstrate that over our 30 years record of data (from 1985 to 2015), the average vegetation cover on mines stays substantially below reference conditions for all mine age classes (figure 2). As such, we cannot empirically estimate a time-to-recovery metric for most sites. In comparison, almost all sites that were deforested but not mined, recovered to reference NDVI values within 20 years (figure 2). Individual sites had high variation in recovery trajectories that likely depend on site specific characteristics beyond the scope of this study.

The LiDAR results largely agree with conclusions from the NDVI metrics, while also highlighting differences in recovery rates within mined portions of the landscapes. There is a significant difference in both the rate of recovery and the starting canopy height between valley fills and other portions of the mined landscape ($p < 0.01$, total mean canopy height growth model $R^2 = 0.98$). At 0 year (i.e. the onset of recovery), valley fills had a mean canopy height of 0.14 m with a growth rate of 0.36 m per year (figures 3(D) and (E)). Non-valley fills start with much higher mean canopy height of 3.3 m and a growth rate of 0.39 m per year (figures 3(D) and (E)). By generously assuming growth rates will continue to be linear, we can estimate how long these two landscape types will take to return to reference mean forest height of 23.3 m. For valley fills it would take ~64 years to return to reference canopy height, and for non-valley fills ~51 years (figure 3(E)). These

rates are slow compared to reported growth rates for the growth of dominant trees in the region's forests, as White Oak and Tulip Poplar, which show a more typical logistic growth rate (figure 3(E); Cotton *et al* 2012).

Canopy height over a 30 m pixel only captures one aspect of the forest cover, so we also calculated canopy closure (as a portion of landscape <2 m tall). Native Appalachian forests have less than 3% of forest canopies in this <2 m category, while even the oldest mined lands (>27 years old) have three times more low stature canopy gaps (>9% of surface) (figure 3(F)). As with mean canopy height, we can model the recovery of forests on mined lands by predicting the decline of areas with <2 m canopy height. Figure 3(F) shows log model equation fits to this decline for both valley fills and non-valley fills, in this case both exhibit an equal decline of shrubland/grassland cover at a rate of 3% per year with the valley fill landscapes starting out with more parts of the landscape with low canopy height ($R^2 = 0.96$ and $p < 0.01$; figure 3(F)). The model estimate suggests that it will take 50 years for non-valley fill landscapes and 61 years for valley fill landscapes to fill in these low canopy heights, results that are consistent with the time scale of recovery from the canopy height dataset. This regional scale analysis of forest recovery is markedly consistent in our focal study of nine local watersheds where our reference watersheds had a mean canopy height of 25 m, 0% shrubland/grassland cover, and a watershed averaged NDVI of 0.85. In comparison, the mined watersheds with mining ages ranging from 0 to 19 years had canopy heights and NDVI recovery rates that were markedly consistent with the regional trends. Unlike at the regional scale however, we had detailed hydrologic and biogeochemical data for these study sites.

3.2. Hydrologic response

Precipitation for the sites ranged from 1050 mm in the RB reference sites to 1100 mm in the BF and SF mined watersheds, potentially due to a small elevation difference between the sites (Nippgen *et al* 2017). Total runoff in the reference watershed was 514 mm. Annual runoff in the mined watersheds MB (0 year), LB (5 years), SF (14 years) and BF (15 years) was 620 mm, 561 mm, 473 mm, and 455 mm, respectively.

Hydrograph separation (2016 data; figure 4) shows an increase in baseflow at all mined sites compared to reference (35% baseflow) with a slightly increased proportion of baseflow in the younger mines (78% and 90% baseflow) compared to the older mines (74% and 68% baseflow). Flow duration curves (figure 5(A)) corroborate these results with increased low flows and lower peak flows at mined sites. These differences do appear to diminish with increasing mine age across this set of sites (figure 5(A)).

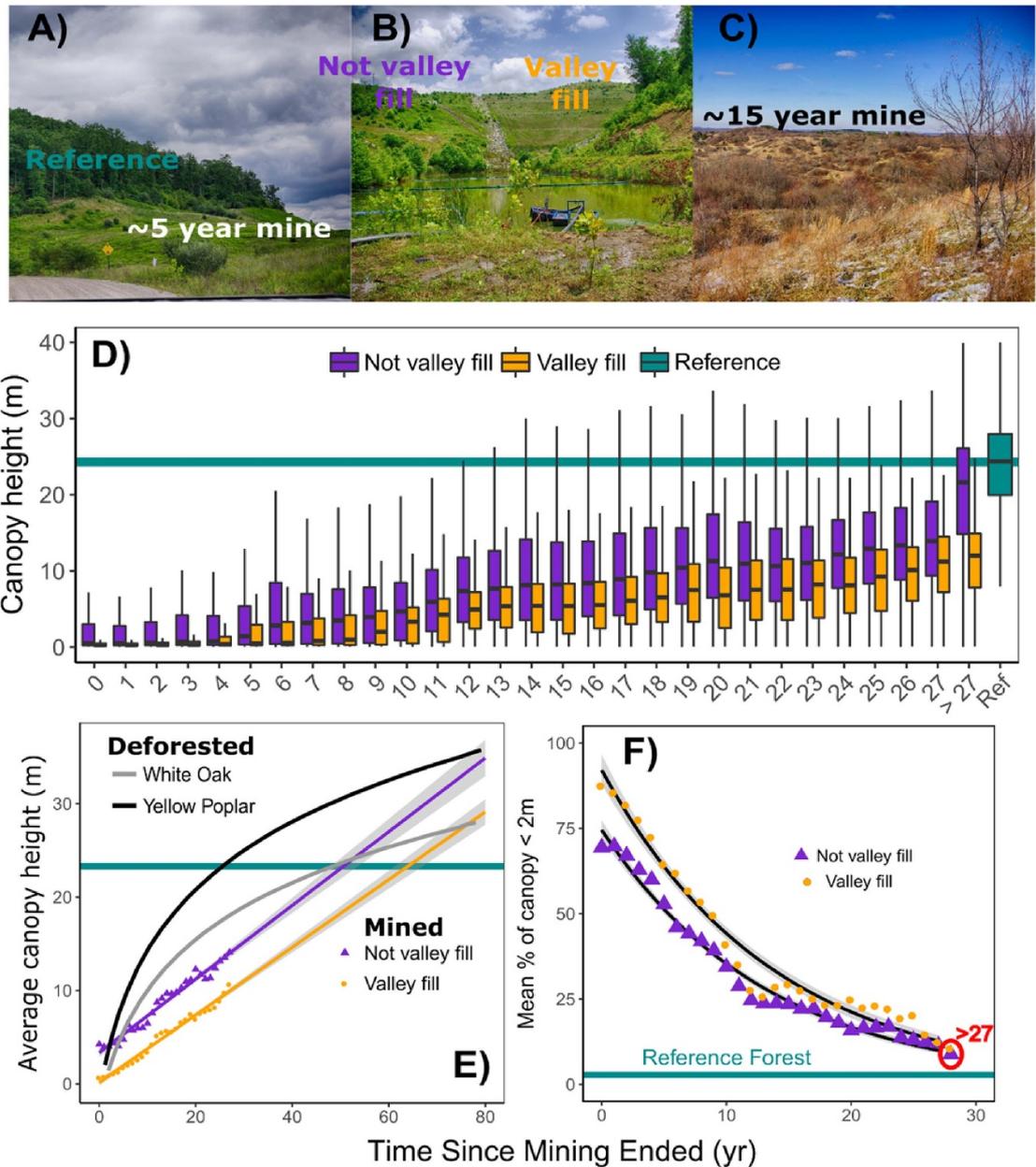
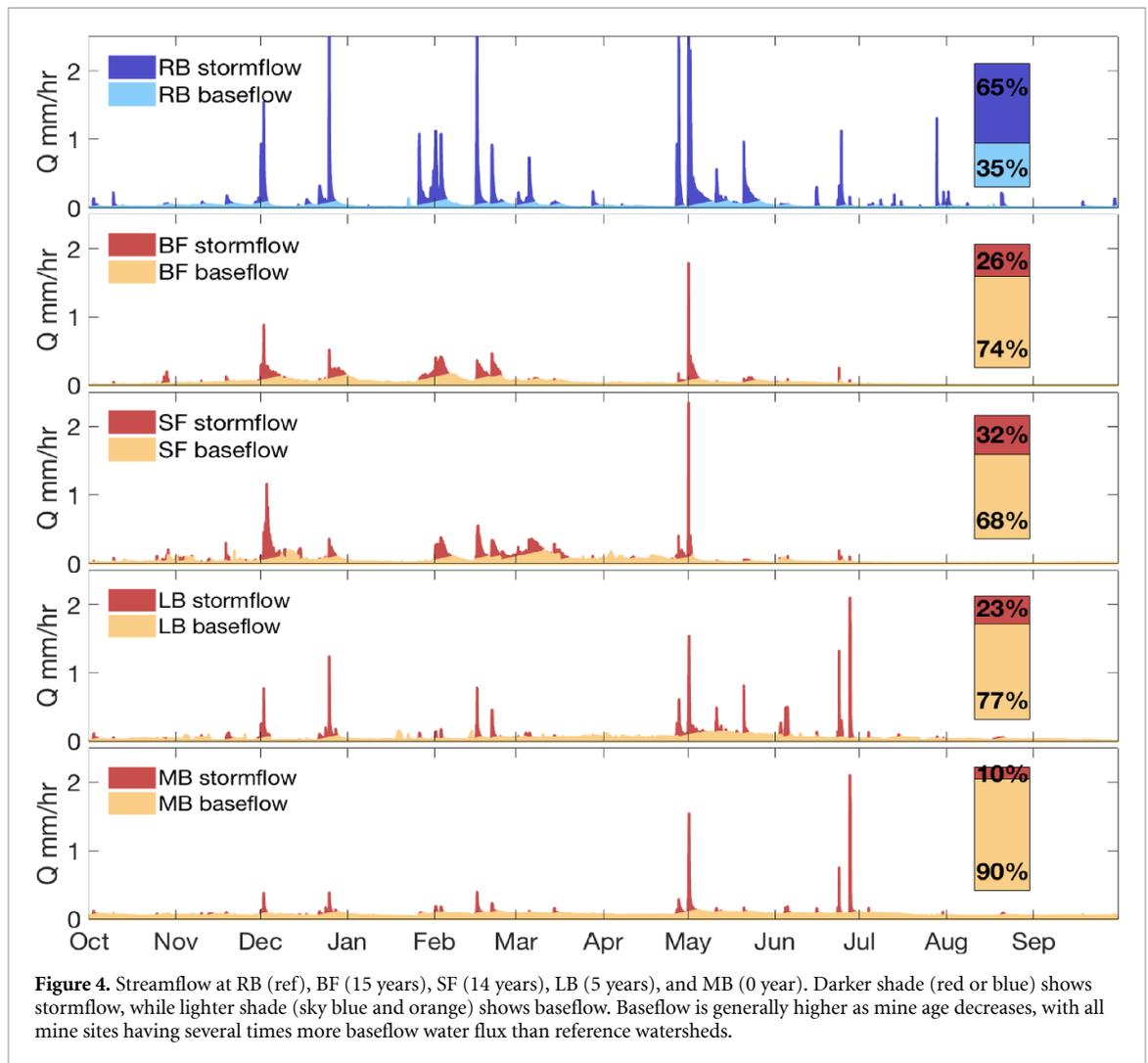


Figure 3. Vegetation change over time in mined, deforested and undisturbed landscape patches. (A) The grass cover on a mined landscape 5 years after mine was reclaimed. (B) The valley fill and toe pond that drain into LB. (C) Vegetation growing on a reconstructed ridge of a mine reclaimed ~15 years previously. (D) Canopy height estimates for vegetation in patches last mined 0 to >27 years earlier are compared to undisturbed forest canopy heights. (E) Trends in reclaimed mine forest regrowth are compared to trends predicted from traditional forestry growth curves for WV forests. (F) The extent of the landscape occupied by vegetation <2 m in height declines with time since mining ceases undisturbed forest patches. In (D)–(F), we compare vegetation recovery between unconsolidated valley fills (yellow) and regraded or recontoured surfaces (purple). Standard deviation of reference canopy height = 0.4 m. Photos by Fabian Nippgen.

Runoff ratios, or the ratio of streamflow to precipitation, were highest at the youngest mine (0.59) and elevated above reference (0.49; figure 5(B)). The 5 years watershed had a runoff ratio near reference at 0.51, while the older mines had runoff ratios below reference at 0.43 and 0.41 for the 14 and 15 years old watersheds. Finally, reference watersheds and sub-watersheds showed a strong decline in zero flow times as watersheds increased in size (figure 5(C)). In contrast, all mined watersheds were perennial and exhibited no zero-flow periods, even for our smallest, 2.4 ha watershed (figure 5(C)).

3.3. Biogeochemical response

Consistent with previous work at these sites (Lindberg *et al* 2011, Nippgen *et al* 2017) stream salinity (measured as SC) was elevated 10–20× above reference conditions (figures 5(D)–(G)) and was less seasonally variable within each mined site (figure 5(D)). This impact is persistent over time, with no consistent decline in median SC for mined sites with increasing mine age when fill volume is accounted for. In contrast, we find that differences in the volume of valley fill can explain nearly all of the variation in median SC across our mined watersheds



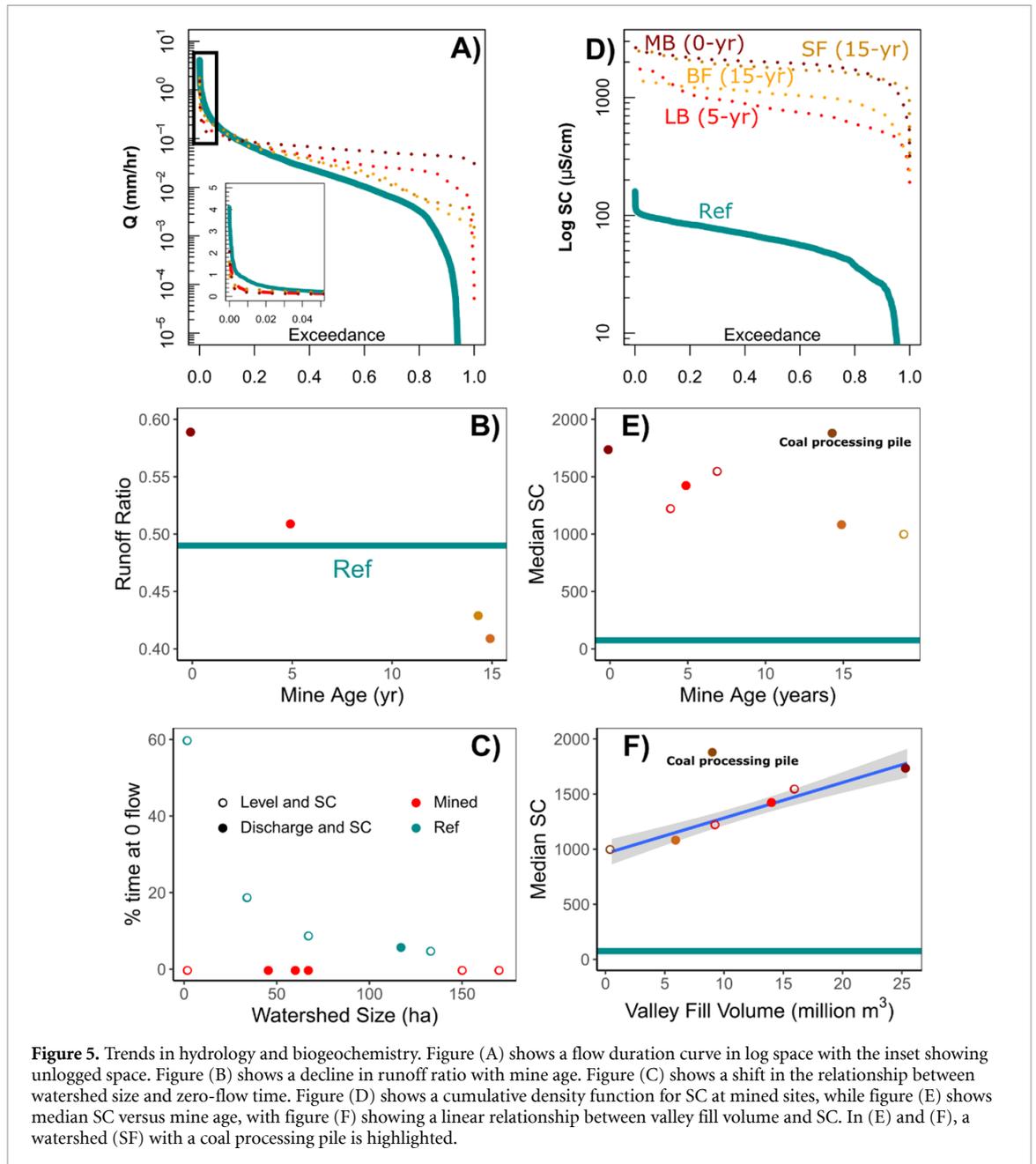
(figure 5(F); $R^2 = 0.93$ and $p < 0.01$). For this regression, we removed one site (SF) that had a coal processing facility at the top of the watershed, which likely further elevated its solute yield above that generated from weathering reactions within fill material.

In addition to simply raising the ion concentration in the streams, MTM also alters weathering element ratios relative to global reference watersheds. While the ion composition of waters in mined watersheds exhibit little variation between mines of different mine age, watershed sizes, and valley fill volume (figure 6), all samples from the seven mined watersheds fall outside of a global average of weathering element ratio of Ca/Na and Mg/Na (figure 6). In contrast, the reference watersheds lie within this global distribution. This shift in composition shows a change in dominant weathering materials after mining and likely represents highly altered soil conditions (figure 6).

4. Discussion

The deep critical zone disturbance caused by MTM creates novel landforms that have fundamentally

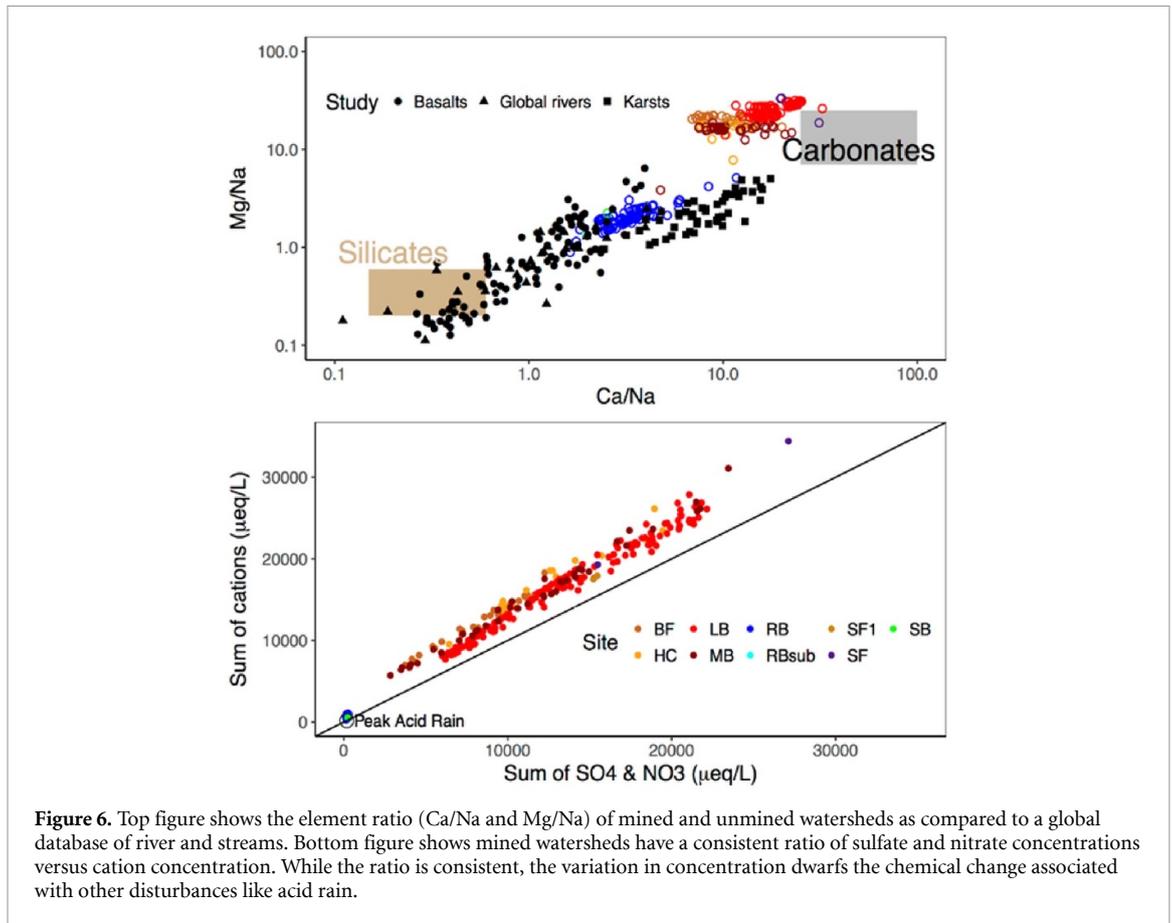
altered ecological, hydrological, and biogeochemical features which will substantially delay or prevent the possibility of forest regrowth. While forest regrowth following traditional timber harvest begin to resemble their pre-disturbance state within 30 years, the vegetation on former mines remains more stunted and sparser than reference forests. The post-mining landscape is less characterized as a forest and more as a semi-open savannah, filled with short-stature non-native trees and grasses, especially on the more heavily engineered valley fills. These landscapes are altered in deep structural ways, creating synthetic landforms with novel geomorphic structures and configuration (Jaeger and Ross 2021). These structural changes fundamentally alter hydrologic and biogeochemical processes that, in part, control ecosystem state (*sensu* Beisner *et al* 2003). As such, the failure to regenerate pre-mining forests and ecosystems may be no surprise. Instead, our study highlights that the legacy of MTM will leave behind potentially permanently altered ecosystems with strongly disrupted connections to their ecological and landscape evolutionary pasts.



4.1. Rebuilt landscapes

Critical zone disruptions in the geology and topography of mined landscapes will leave a permanent legacy on the shape of the land in Central Appalachia. Mining operations have deepened the functional critical zone from typically less than 2 m to bedrock (Sucre *et al* 2011) to 100 ms (Ross *et al* 2016), creating novel landforms (Maxwell and Strager 2013) with flatter topography (Ross *et al* 2016) and altered, land surface energy balances due to changes in slope and aspect (Wickham *et al* 2013). These foundational alterations to the critical zone are similar to volcanic eruptions in their ability to completely reset landscape, ecosystem, and hydrologic co-evolution (Ross *et al* 2016). On top of and underneath these novel landforms, there are fundamentally new controls and constraints on vegetation recovery and hydrologic

and biogeochemical cycling. Together, these changes highlight that post-MTM landscapes are unlikely to ‘recover’ in a traditional way, rather they have become novel ecosystems (Hobbs *et al* 2009) on top of novel landforms. Current regulations including SMCRA that encourage a classic secondary-succession based restoration approach in such heavily altered landscapes are inappropriate and unlikely to succeed, as novel ecosystems require novel management approaches (Seastedt *et al* 2008). In these designed landscapes, successful recovery after mining may be working towards different design goals (Ross *et al* 2015). Novel ecosystem design projects are already in progress in the region, including rewilding efforts that reintroduce Elk (*Cervus canadensis*) to the post-mining grasslands (Lituma *et al* 2021) and the establishment of extensive solar and wind



power infrastructure in treeless landscapes (Skousen and Zipper 2021).

4.2. Constraints on ecological recovery on mined landscapes

Mined landscapes can be conceptualized as a hybrid between primary ecological succession onto freshly exposed bedrock or secondary succession into grassland ecosystems that have been planted by mine operators. In the primary succession case, where mine spoil is analogous to glacial till or fresh volcanic ash, mined landscape forest growth and productivity are limited by classic primary succession factors like limited seed dispersal (Fastie 1995), nutrient limitation (Vitousek *et al* 1993), and little or no soil (Moral 2007). Previous research on mined landscapes has found that these primary succession factors do limit forest growth and native plant recovery on mined landscapes with research showing seed dispersal limitations to some native species (Holl *et al* 1994), nutrient and organic matter additions improving tree growth (Lindsay *et al* 2013), and low soil organic matter on mine lands negatively impacting vegetation regrowth (Acton *et al* 2011).

In contrast, there is also abundant research that the primary limits to forest regrowth are more like secondary succession in nature, with competition,

herbivory, and shading acting as primary limits to forest regrowth (Torbert and Burger 1990, Torbert *et al* 1995, Zipper *et al* 2011, Franklin *et al* 2012). However, the two cases are not mutually exclusive as post-mining landscapes can contain areas of no soil or vegetation and areas with heavy replanting and soil amendments (Wunsch *et al* 1999, Nash *et al* 2016). In our dataset, there is a clear difference between valley-fills and non-valley fills, which could be related to these two modes of succession. Valley fill benches are heavily compacted and planted with erosion preventing grasses and shrubs, and they start out with lower canopies than non-valley fills. This height difference persists for >25 years, even though valley fills typically have more edge contact with intact forests in the former valley bottoms. Previous research has suggested that this persistent lack of forest may be due to competitive interactions between herbaceous cover, invasive and planted shrubs, and trees (Franklin *et al* 2012) and soil compaction (Phelps *et al* 1981). In contrast, non-valley fill areas are more typically directly seeded with forest species, often directly into a spoil/soil mix (Zipper *et al* 2011), and our data show they start out with a higher canopy height. Slow recovery on these sites may be less due to secondary succession factors like competition, and more to do with poor primary succession conditions like spoil/soil health (Lindsay *et al* 2013, Nash *et al* 2016).

The persistent changes in forest canopy structure of lower canopy height, more open areas, and lower stem density will have long-term impacts on the biodiversity of the region given such extensive mining (10% of the landscape). For example, canopy height and architecture are known to positively correlate with biodiversity by increasing the three-dimensional volume of habitat space, especially for birds (MacArthur and Pianka 1966, Goetz *et al* 2010). While mountaintop mined areas do not show a decrease in the total number of bird species, prior surveys have documented that bird species shift from dominance by native forest birds to dominance by species typically reported from Midwestern grasslands (EPA 2005). There is great concern that the loss of contiguous Appalachian forest and its replacement with persistent grassland/shrubland habitat is threatening endangered Appalachian forest birds like the Cerulean Warbler (Wickham *et al* 2007, Becker *et al* 2014).

4.3. Longevity of downstream impacts of mountaintop mines on the hydrology and chemistry of receiving streams

In previous studies of forest disturbance by clear-cutting, forest recovery leads to hydrologic and biogeochemical cycles that tend towards reference conditions as forest biomass accumulates (Likens *et al* 1970, Swank and Douglass 1974, Johnson *et al* 1988, Adams *et al* 1994, Sahin and Hall 1996). In the case of MTM recovery, we do observe strong coupling between vegetation and hydrology, with declines in water yield as rates of evapotranspiration increase with vegetation cover. These water yield decreases, however, do not return to pre-mining hydrology. The streams draining our oldest mines still have sustained, higher baseflows and lower runoff ratios. The most likely explanation for this phenomenon is that mining operations have created significant additional groundwater storage within the spoil material (Nippgen *et al* 2017). Given the depth of valley fills, it is likely that a large portion of this stored groundwater is inaccessible to plants, but readily discharged to streams, as no mined sites had zero-flow days despite the reduction in runoff ratio. These changes show a radically different partitioning of water in the landscape with enhanced baseflows and lowered runoff ratios, potentially indicating more evapotranspiration (despite lower biomass) or deeper groundwater losses.

The biogeochemistry of this deep pool of water stored 10 ms below the surface is not controlled by terrestrial ecosystem nutrient cycling (Likens 2001). Instead, the chemistry of streamflow remains controlled by strong acid weathering of pyrite minerals in a matrix of carbonate minerals (Ross *et al* 2018). These reactions generate alkaline mine drainage with elevated ion and pollutant concentrations for at least decades (Ross *et al* 2016), but likely longer given the

complete decoupling of surface vegetation and stream biogeochemistry.

5. Conclusions

MTM is not only an ecological disturbance on the land surface, it is a form of disturbance that disrupts and reorganizes the entire critical zone, fundamentally altering the landscape in ways that alter all environmental processes from geological to ecological. As such, ecological restoration approaches like those required by law have largely failed to restore pre-mining forests to post-mining landscapes throughout WV. The legacy of MTM covers over 6000 km² of land with novel ecosystems of open canopy savannas, streams that never go dry as they emerge from 100 ms of spoil piles, and elevated chemical weathering rates that are among the fastest in the world. These changes interact to make a new system, one that is mostly unrelated to how it was before. The ecologic, hydrologic, and biogeochemical processes active in these new places last for at least 30 years and may be on new trajectories of evolution that will keep them different from unmined landscapes forever. These systems can still provide value for people and nature, but they do not meet SMCRA restoration goals, and restoration itself is likely a doomed metaphor for creating value once the coal is gone.

Data availability statement

The data that support the findings of this study are available upon reasonable request from the authors.

Acknowledgments

This research was funded by a National Science Foundation EAR Hydrologic Sciences Grant 1417405 to B L McGlynn and E S Bernhardt and an NSF Graduate Research Fellowship to M Ross. The staff of WV DNR District 5 Upper Mud River provided logistical support with special thanks to Nick Huffman. We would like to thank Anita and Stanely Miller for property access and support in the field. All data can be found here: <https://doi.org/10.6084/m9.figshare.c.5437662>.

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References

- Acton P M, Fox J F, Campbell J E, Jones A L, Rowe H, Martin D and Bryson S 2011 Role of soil health in maintaining environmental sustainability of surface coal mining *Environ. Sci. Technol.* **45** 10265–72

- Adams M B, Kochenderfer J N, Wood F, Angradi T R and Edwards P 1994 *Forty Years of Hydrometeorological Data from the Fernow Experimental Forest, WV* 184
- Becker D A, Wood P B, Strager M P and Mazzarella C 2014 Impacts of mountaintop mining on terrestrial ecosystem integrity: identifying landscape thresholds for avian species in the central Appalachians, United States *Landsc. Ecol.* **30** 339–56
- Beisner B E, Haydon D T and Cuddington K 2003 Alternative stable states in ecology *Front. Ecol. Environ.* **1** 376–82
- Brooks A C, Ross M R V, Nippgen F, McGlynn B L and Bernhardt E S 2019 Excess nitrate export in mountaintop removal coal mining watersheds *Wiley Online Library* **124** 3867–80
- Clements F 1916 *Plant Succession: An Analysis of the Development of Vegetation* 242 (Carnegie Institution of Washington)
- Congress U S 1977 The surface mining control and reclamation act (SMCRA) of 1977
- Cotton C, Barton C, Lhotka J, Angel P N and Graves D 2012 Evaluating reforestation success on a surface mine in Eastern Kentucky *USDA For. Serv. Proc.* vol **RMRS-P-68** pp 16–23
- Cowles H C 1911 The causes of vegetational cycles *Ann. Assoc. Am. Geogr.* **1** 3–20
- de Richter D B and Billings S A 2015 ‘One physical system’: Tansley’s ecosystem as Earth’s critical zone *New Phytol.* **206** 900–12
- Dicken C L, Nicholson S W, Horton J D, Foose M P and Mueller J A L 2005 Preliminary integrated geologic map databases for the United States: AL, FL, GA, MS, NC, and SC *US Geol. Surv. Open-File Rep.* 2005-1323 vol 1323
- EPA 2005 Mountaintop mining/valley fills in Appalachia final programmatic environmental impact statement (available at: [papers2://publication/uuid/02D18A2E-5769-43B0-A0CB-6C8EEF625061](https://pubs2://publication/uuid/02D18A2E-5769-43B0-A0CB-6C8EEF625061))
- Evans D M, Zipper C E, Hester E T and Schoenholtz S H 2015 Hydrologic effects of surface coal mining in Appalachia (US) *JAWRA J. Am. Water Resour. Assoc.* **24061** n/a
- Fastie C 1995 Causes and ecosystem consequences of multiple pathways of primary succession at Glacier Bay, AK *Ecology* **76** 1899–916
- Franklin J A, Zipper C E, Burger J A, Skousen J G and Jacobs D F 2012 Influence of herbaceous ground cover on forest restoration of eastern US coal surface mines *New For.* **43** 905–24
- Goetz S J, Steinberg D, Betts M G, Holmes R T, Doran P J, Dubayah R and Hofton M 2010 LIDAR remote sensing variables predict breeding habitat of a neotropical migrant bird *Ecology* **91** 1569–76
- Greer B M, Burbey T J, Zipper C E and Hester E T 2017 Electrical resistivity imaging of hydrologic flow through surface coal mine valley fills with comparison to other landforms *Hydrol. Process* **31** 2244–60
- Hobbs R J, Higgs E and Harris J A 2009 Novel ecosystems: implications for conservation and restoration *Trends Ecol. Evol.* **24** 599–605
- Holl K D, Cairns J and Cairns J 1994 Vegetational community development on reclaimed coal surface mines in VA *Bull. Torrey Bot. Club* **121** 327–37
- Hooke R L B, Martín-Duque J F and Pedraza J 2012 Land transformation by humans: a review *GSA Today* **22** 4–10
- Hurttt G C, Dubayah R, Drake J, Moorcroft P R, Pacala S W, Blair J B and Fearon M G 2004 Beyond potential vegetation: combining LIDAR data and a height-structured model for carbon studies *Ecol. Appl.* **14** 873–83
- Jaeger K L and Ross M R V 2021 Identifying geomorphic process domains in the synthetic landscapes of WV, USA *J. Geophys. Res. Earth Surf.* **126** 1–19
- Jenny H 1994 *Factors of Soil Formation: A System of Quantitative Pedology* (New York: McGraw-Hill)
- Johnson D, Kelly J, Swank W T, Cole D W, van Miergroet H, Hornbeck J W, Pierce R S and van Lear D 1988 The effects of leaching and whole-tree harvesting on cation budgets of several forests *J. Environ. Qual.* **17** 418–24
- Likens G E 2001 Biogeochemistry, the watershed approach: some uses and limitations *Mar. Freshw. Res.* **52** 5
- Likens G, Bormann F, Noye M, Fisher D and Pierce R 1970 Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard brook watershed-ecosystem *Ecol. Monogr.* **40** 23–47
- Lindberg T T, Bernhardt E S, Bier R, Helton A M, Merola R B, Vengosh A and Di Giulio R T 2011 Cumulative impacts of mountaintop mining on an Appalachian watershed *Proc. Natl Acad. Sci. USA* **108** 20929–34
- Lindsay W-K, Curtis D, Calene T, Paul E, Keith O and Jeff S 2013 Hardwood tree growth on amended mine soils in WV *J. Environ. Qual.* **42** 1363–71
- Lituma C M, Cox J J, Spear S F, Edwards J W, de la Cruz J L, Muller L I and Ford W M 2021 Terrestrial wildlife in the post-mined Appalachian landscape: status and opportunities *Appalachia’s Coal-Mined Landscapes* (Cham: Springer) pp 135–66
- MacArthur R H and Pianka E R 1966 On optimal use of a patchy environment *Am. Nat.* **100** 603–9
- Maxwell A E and Strager M P 2013 Assessing landform alterations induced by mountaintop mining *Nat. Sci.* **05** 229–37
- Maxwell A E, Strager M P, Yuill C B and Petty J T 2012 Modeling critical forest habitat in the southern coal fields of WV *Int. J. Ecol.* **2012** 1–10
- McGaughey R 2012 FUSION/LDV: software for LIDAR data analysis and visualization. Version 3.10. USDA For. Serv. Pacific Northwest Res. Station. Seattle, WA (available at: <https://scholar.google.com/scholar?q=FUSION%2FLDV%3A+Software+for+LIDAR+data+analysis+and+visualization.+Version+3.10.+USDA+For.+Serv&btnG=&hl=en&sdt=0%2C34>)
- Moral D 2007 Limits to convergence of vegetation during early primary succession *J. Veg. Sci.* **18** 479–88
- Nash W L, Daniels W L, Haering K C, Burger J A and Zipper C E 2016 Long-term effects of rock type on Appalachian coal mine soil properties *J. Environ. Qual.* **45** 1597–606
- Naslund L C, Gerson J R, Brooks A C, Walters D M and Bernhardt E S 2020 Contaminant subsidies to riparian food webs in Appalachian streams impacted by mountaintop removal coal mining *Environ. Sci. Technol.* **54** 3951–9
- Nippgen F, Ross M R V, Bernhardt E S and McGlynn B L B L 2017 Creating a more perennial problem? Mountaintop removal coal mining enhances and sustains saline base flows of Appalachian watersheds *Environ. Sci. Technol.* **51** 8324–34
- Odum E P 1969 The strategy of ecosystem development *Science* **164** 262–70
- Oosting H J 1942 An ecological analysis of the plant communities of piedmont, NC *Am. Midl. Nat.* **28** 1
- Pericak A A, Thomas C J, Kroodsma D A, Wasson M F, Ross M R V, Clinton N E, Campagna D J, Franklin Y, Bernhardt E S and Amos J F 2018 Mapping the yearly extent of surface coal mining in central appalachia using Landsat and Google earth engine *PLoS One* **13** 1–15
- Phelps L B, Wells W B Jr and Saperstein L W 1981 Analysis of surface coal mine spoil bulk density *Preprint—Society of Mining Engineers of AIME* (Littleton, CO: Soc of Min of AIME)
- Reed M and Kite S 2020 Peripheral gully and landslide erosion on an extreme anthropogenic landscape produced by mountaintop removal coal mining *Earth Surf. Process. Landf.* **45** 2078–90
- Rodrigue J A, Burger J A and Oderwald R G 2002 Forest productivity and commercial value of pre-law reclaimed mined land in the eastern United States *North. J. Appl. For.* **19** 106–14

- Ross M R V, Bernhardt E S, Doyle M W and Heffernan J B 2015 Designer ecosystems: incorporating design approaches into applied ecology *Annu. Rev. Environ. Resour.* **40** 419–43
- Ross M R V, Nippgen F, Hassett B A, McGlynn B L and Bernhardt E S 2018 Pyrite oxidation drives exceptionally high weathering rates and geologic CO₂ release in mountaintop-mined landscapes *Glob. Biogeochem. Cycles* **32** 1182–94
- Ross M R V, Nippgen F, McGlynn B L and Bernhardt E S 2016 Deep impact: effects of mountaintop mining on surface topography, bedrock structure, and downstream waters *Environ. Sci. Technol.* **50** 2064–74
- Roy D P, Kovalsky V, Zhang H K, Vermote E F, Yan L, Kumar S S and Egorov A 2016 Characterization of Landsat-7 to Landsat-8 reflective wavelength and normalized difference vegetation index continuity *Remote Sens. Environ.* **185** 57–70
- Sahin V and Hall M J 1996 The effects of afforestation and deforestation on water yields *J. Hydrol.* **178** 293–309
- Schoenholtz S H, Burger J A and Torbert J L 1987 Natural mycorrhizal colonization of pines on reclaimed surface mines in VA *J. Environ. Qual.* **16** 143–6
- Seastedt T R, Hobbs R J and Suding K N 2008 Management of novel ecosystems: are novel approaches required? *Front. Ecol. Environ.* **6** 547–53
- Skousen J, Gorman J, Pena-Yewtukhiw E, King J, Stewart J, Emerson P and DeLong C 2009 Hardwood tree survival in heavy ground cover on reclaimed land in WV: mowing and ripping effects *J. Environ. Qual.* **38** 1400–9
- Skousen J and Zipper C E 2021 *Coal Mining and Reclamation in Appalachia Appalachia's Coal-Mined Landscapes* (Springer) pp 55–83
- Sucre E, Tuttle J and Fox T 2011 The use of ground-penetrating radar to accurately estimate soil depth in rocky forest soils *For. Sci.* **57** 59–66
- Swank W T and Douglass J E 1974 Streamflow greatly reduced by converting deciduous hardwood stands to pine *Science* **185** 451–2
- Team R C 2013 A language and environment for statistical computing 201
- Torbert J L and Burger J A 1990 Tree survival and growth on graded and ungraded minesoil *Tree Plant. Notes* **41** 3–5
- Torbert J L, Burger J A and Probert T 1995 Evaluation of techniques to improve white pine establishment on an Appalachian minesoil *J. Environ. Qual.* **24** 869–73
- Vitousek P M, Walker L R and Whiteaker L D 1993 Nutrient limitations to plant growth during primary succession in Hawaii volcanoes national park *Biogeochemistry* **23** 197–215
- Wickham H *et al* 2019 Welcome to the tidyverse *J. Open Source Softw.* **4** 1686
- Wickham J D, Riitters K H, Wade T G, Coan M and Homer C 2007 The effect of Appalachian mountaintop mining on interior forest *Landsc. Ecol.* **22** 179–87
- Wickham J, Wood P B, Nicholson M C, Jenkins W, Druckenbrod D, Suter G W, Strager M P, Mazzarella C and Amos J 2013 The overlooked terrestrial impacts of mountaintop mining *Bioscience* **63** 335–48
- Wright J P and Fridley J D 2010 Biogeographic synthesis of secondary succession rates in eastern North America *J. Biogeogr.* **37** 1584–96
- Wunsch D R, Dinger J S and Graham C D R 1999 Predicting ground-water movement in large mine spoil areas in the Appalachian plateau *Int. J. Coal Geol.* **41** 73–106
- Zegre N P, Miller A J, Maxwell A and Lamont S J 2014 Multiscale analysis of hydrology in a mountaintop mine-impacted watershed *JAWRA J. Am. Water Resour. Assoc.* **50** 1257–72
- Zhao F *et al* 2018 Development of Landsat-based annual US forest disturbance history maps (1986–2010) in support of the North American carbon program (NACP) *Remote Sens. Environ.* **209** 312–26
- Zipper C E, Burger J A, Skousen J G, Angel P N, Barton C D, Davis V and Franklin J A 2011 Restoring forests and associated ecosystem services on Appalachian coal surface mines *Environ. Manage.* **47** 751–65