Evaluating expected outcomes of acid remediation in an intensively mined Appalachian watershed

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Received: 15 November 2016 / Accepted: 30 May 2017 © Springer International Publishing Switzerland 2017

Abstract Assessments of watershed-based restoration efforts are rare but are essential for the science of stream restoration to advance. We conducted a watershed scale assessment of Abram Creek before and after implementation of a watershed-based plan designed to maximize ecological recovery from acid mine drainage (AMD) impairment. We surveyed water chemistry, physical habitat, benthic macroinvertebrates, and fish community structure in three stream types: AMD-impacted (14 streams), AMD-treated (13 streams), and unimpaired reference (4 streams). We used in-stream measurements to quantify ecological loss from AMD, the amount of ecological recovery expected through remediation, and the observed degree of post-treatment recovery. Sites impaired by AMD improved in water quality with AMD treatment. Dissolved metals and acidity declined significantly in treated streams, but sulfate and specific conductance did not. Likewise, sites impaired by AMD improved in bio-condition scores with AMD treatment. EPT genera increased significantly but were lower compared to unimpaired streams. We found fish at nine treated sites that had none before treatment. Community-level analyses indicated improved but altered assemblages with AMD treatment. Analysis of pre-treatment conditions indicated that only 30% of the historic fishery remained. Remediation was expected to recover 66% of the historic fishery value, and assessment of post-treatment conditions indicates that 52% of the historic fishery has been recovered after 3 years. Developing expected endpoints for restoration outcomes provides a tool to objectively evaluate successes and can guide adaptive management strategies.

Keywords Acid mine drainage · Acid remediation · Adaptive watershed management · Ecological units · Watershed scale remediation · Predictive models

Introduction

Stream restoration is increasingly used to alleviate negative impacts associated with anthropogenic landscape change. In the USA alone, an average of $1 billion dollars per year is invested in this effort (Bernhardt et al. 2005). Surprisingly, however, there are no agreed upon standards for what constitutes success given this large investment (Palmer et al. 2005) and assessment of such projects are rare (Bernhardt et al. 2005; Heinrich et al. 2014). In central Appalachia, a considerable amount of research and development targets remediation of acid-impacted watersheds (Freund and Petty 2007; Merovich et al. 2007; Petty et al. 2008). In this region, coal has been mined for
nearly 200 years (Merovich et al. 2007), and as a consequence, over 17,000 km of streams in the Mid-Atlantic Highlands are impacted by acid mine drainage (AMD) (USEPA 2000). In fact, the legacy effects of coal mining on aquatic ecosystems represent a crucial environmental issue in this region (Petty et al. 2010).

Historically, most restoration projects were small scale (Bernhardt et al. 2005) and focused on improving local (stream segment-level) environmental conditions. Localized projects commonly fall victim to the “field of dreams” myth (Hilderbrand et al. 2005). This myth, of ecological restoration, is based on the assumption that if the habitat is restored, the function of the ecosystem will self-assemble along with the recolonization of biotic communities (Hilderbrand et al. 2005). This assumption that “if you build it they will come” ignores processes occurring in the broader, watershed scale context (Merovich et al. 2013). In fact, metacommunity theory suggests that “if you build it, they may not come” (Brown et al. 2011). Consequently, remediation programs must focus on recovering connectivity among stream reaches within the entire drainage network (Jansson et al. 2007; Lake et al. 2007; McClurg et al. 2007). This approach requires addressing a strategic proportion of identified problem sources at the watershed scale (Petty et al. 2008).

The purpose of this paper is to present a case where a strategic watershed-based remediation initiative was implemented around AMD treatment to maximize watershed scale recovery. We ask: what is the expected benefit of the initiative and how does it compare to observed outcomes? We studied Abram Creek, a sub-watershed of the North Branch of the Potomac River basin in the eastern panhandle of West Virginia (WV). We answered the question with a unique analysis of data collected around a watershed-based experimental survey designed specifically to (1) quantify the response of water chemistry, benthic macroinvertebrates, and fish to AMD treatment 3 years after remediation efforts were implemented and (2) evaluate the predicted outcomes of remediation for different aquatic life end points. This effort to address the expected outcomes of acid remediation on aquatic life at the watershed scale is the first we know of, and it could provide critical information needed in adaptive management framework for managing heavily impacted watersheds (Petty et al. 2010; Merovich et al. 2013).

Methods

Study area

Abram Creek is a 115-km² watershed (North Branch/Potomac River basin) located in the eastern panhandle of WV in Grant and Mineral Counties (latitude 39° 18′ 44.8″, longitude 79° 12′ 41.2″). The Abram Creek mainstem is 31.5 km and flows north from an elevation of 1065 m in the headwaters to 516 m at its confluence with the North Branch of the Potomac River upstream of Kitzmiller, Maryland (Fig. 1). Land cover is dominated by forest (66%) and agriculture (25%); geology consists predominantly of shale and sandstone. The watershed contains 23 mapped stream segments; the largest of which are Emory Creek, Glade Run, Johnnycake Run, and Laurel Run (Fig. 1). Impairments throughout the watershed are primarily due to AMD from abandoned mine lands (AMLs; West Virginia Water Research Institute (WVWRI) 2007). West Virginia Department of Environmental Protection (WVDEP), Division of Water and Waste Management identified 27 abandoned mine sources (discharges, seeps, portals, culverts, refuse piles, diversion ditches, and ponds) throughout the study area (WVWRI 2007). There are eight National Pollution Discharge Elimination System (NPDES) permits in the watershed for metal effluents related to mining (WVWRI 2007).

In 2007, the WVWRI proposed a strategic watershed-based remediation plan, following adaptive watershed management principles, for the Abram Creek watershed to the WVDEP Office of Abandoned Mine Land and Reclamation. In 2010, the project was completed. The AMD treatment technology implemented in the watershed includes three liming dosers at the Abram Creek headwaters: Abram Creek right fork, Little Creek, and an unnamed tributary at river kilometer 6.2; two limestone sand dump sites at Laurel Run and Emory Creek; and one passive treatment system at Glade Run (Fig. 1 and Table 1). Dosers were installed adjacent to impacted tributaries. This treatment technology diverts water to a water wheel that drives an auger adding calcium oxide in proportion needed to neutralize in-stream acidity. Treated water returns to the stream where precipitation reactions occur. Limestone sand application is performed by dumping sand along the stream bank where gravity and stream water gradually wash the sand downstream, thereby neutralizing acidity (McClurg et al. 2007). The passive treatment system
installed incorporated kiln dust, a waste product of the limestone industry, into the stream bed and bank. Acidic water from abandoned mine portals passes over this material and is neutralized.

Site selection

We studied 18 sites within the study area (Table 1). Sites were strategically selected based on treatment locations to evaluate the putative benefits of treatment (Fig. 1). In 2008, 13 AMD sites and 1 reference site were sampled to quantify pre-treatment conditions. These same 14 sites were sampled again in 2013 to quantify post-treatment conditions, along with 4 additional sites to increase the number of unimpaired reference and untreated AMD streams (Table 1).

Streams were classified into three a priori types: (1) streams impaired by AMD (14 streams), (2) streams treated for AMD impairment (13 streams), and (3) unimpaired reference streams (4 streams) (Table 1). AMD (A) streams were listed on the 2004 303d list for water quality and/or biological impairment (WVWRI 2007). Treated (T) streams received AMD treatment or were downstream of treatment. Reference (R) streams were naturally circumneutral. Circumneutral streams within the study area represent the best available conditions in the watershed and provide the only reasonable reference condition against which to assess watershed-based remediation plans (Campbell 2000, McClurg et al. 2007).

Data collection

We monitored water chemistry in May of 2008 and 2013 at each assessment site by collecting a 1-L unfiltered grab sample and 500-mL filtered sample (0.45-μm pore, mixed cellulose ester membrane discs) (see
Merovich and Petty 2007). Samples were analyzed at the National Research Center for Coal and Energy at West Virginia University for alkalinity (mg/L CaCO$_3$ equivalents), acidity (mg/L CaCO$_3$ equivalents), SO$_4^{2-}$ (mg/L), and dissolved Al, Ba, Cu, Cl$^-$, Co, Cr, Cd, Ca, Na, Ni, Se, Zn, Fe, Mg, and Mn concentrations (mg/L). We also measured temperature (°C), pH, specific conductance (μS/cm), dissolved O$_2$ (mg/L), and total dissolved solids (g/L) in situ using a YSI 650 with a 600 XL sonde (Yellow Springs Instruments, Yellow Springs, OH). In addition, we evaluated physical habitat condition with rapid visual habitat assessment (RVHA) techniques for each site in the late summer of 2008 and 2013 following US Environmental Protection Agency (USEPA) protocols (Barbour et al. 1999).

We monitored biological condition at each site by collecting benthic macroinvertebrate and fish assemblage data. Benthic macroinvertebrates were sampled at each site in May of 2008 and 2013 following rapid bioassessment protocols for wadeable rivers (Barbour et al. 1999). We obtained four kick samples at each site using a rectangular kick-net (net dimensions 355 × 508 mm with 500 μm netting) from widely separated riffle habitat to sample a total of 1.0 m$^2$. We filtered all four samples through a 250-μm sieve and combined them into a single composite sample. The composite sample was immediately preserved in 95% ethanol. Subsampling followed a modified version of the USEPA’s Rapid Bioassessment Protocol. We selected a sub-sample of 200 macroinvertebrates by picking individuals from randomly selected grid cells (WVDEP 2013). We identified all macroinvertebrates to genus or the lowest possible taxonomic level using Peckarsky et al. (1990) and Merritt and Cummins (2008). We collected fish assemblage data in the late summer of 2008 and 2013 using one to three backpack electrofishing units (Smith Root models 12-B, 15-D, and/or LR-24) depending on the size of the stream (Freund and Petty 2007). Reach lengths were 40 times the mean stream width with a minimum of 150 m and a maximum of 300 m (Freund and Petty 2007). We used single passes, which have been found to be sufficient at these distances for estimating relative species richness or comparing relative biological conditions (Freund and Petty 2007). All individual fish captured were identified to species, counted, and returned to the stream alive.

### Statistical analyses

Our first objective was to quantify the response of water chemistry, benthic macroinvertebrates, and fish to AMD treatment 3 years post-treatment. To meet this objective,

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**Table 1** Site names, GPS coordinates of sampling locations, stream type, and treatment technology implemented

<table>
<thead>
<tr>
<th>Site name</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Stream type 2008</th>
<th>Stream type 2013</th>
<th>Treatment type</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abram Creek at mouth</td>
<td>39.37938</td>
<td>−79.20199</td>
<td>AMD (A1)</td>
<td>Treated (T1)</td>
<td>Downstream</td>
</tr>
<tr>
<td>Abram Creek above Emory Creek</td>
<td>39.35369</td>
<td>−79.17154</td>
<td>AMD (A3)</td>
<td>Treated (T3)</td>
<td>Downstream</td>
</tr>
<tr>
<td>Emory Creek at mouth</td>
<td>39.35429</td>
<td>−79.16722</td>
<td>AMD (A2)</td>
<td>Treated (T2)</td>
<td>LS sand</td>
</tr>
<tr>
<td>Unnamed tributary 2 of Emory Creek</td>
<td>39.33565</td>
<td>−79.15524</td>
<td>NS</td>
<td>Reference (R2)</td>
<td>–</td>
</tr>
<tr>
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<td>39.33565</td>
<td>−79.15599</td>
<td>NS</td>
<td>Reference (R3)</td>
<td>–</td>
</tr>
<tr>
<td>Emory Creek headwater left fork</td>
<td>39.33565</td>
<td>−79.15524</td>
<td>NS</td>
<td>AMD (A14)</td>
<td>–</td>
</tr>
<tr>
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<td>39.35058</td>
<td>−79.18403</td>
<td>AMD (A4)</td>
<td>Treated (T4)</td>
<td>Downstream</td>
</tr>
<tr>
<td>Johnnycake Run at mouth</td>
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<td>−79.21424</td>
<td>Reference (R)</td>
<td>Reference (R)</td>
<td>–</td>
</tr>
<tr>
<td>Upper Johnnycake Run</td>
<td>39.30171</td>
<td>−79.21109</td>
<td>NS</td>
<td>Reference (R4)</td>
<td>–</td>
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<td>39.31370</td>
<td>−79.21385</td>
<td>AMD (A5)</td>
<td>Treated (T5)</td>
<td>Downstream</td>
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<td>Glade Run at mouth</td>
<td>39.30629</td>
<td>−79.18667</td>
<td>AMD (A6)</td>
<td>Treated (T6)</td>
<td>Passive</td>
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<td>Abram Creek above Glade Run</td>
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<td>−79.18884</td>
<td>AMD (A7)</td>
<td>Treated (T7)</td>
<td>Downstream</td>
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<td>−79.19072</td>
<td>AMD (A8)</td>
<td>Treated (T8)</td>
<td>LS sand</td>
</tr>
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<td>Abram Creek above Laurel Run</td>
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<td>−79.19087</td>
<td>AMD (A9)</td>
<td>Treated (T9)</td>
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<td>Abram Creek at Vindex</td>
<td>39.23752</td>
<td>−79.21071</td>
<td>AMD (A10)</td>
<td>Treated (T10)</td>
<td>Downstream</td>
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<td>Abram Creek at CR 42</td>
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<td>−79.21660</td>
<td>AMD (A11)</td>
<td>Treated (T11)</td>
<td>Downstream</td>
</tr>
<tr>
<td>Little Creek</td>
<td>39.21851</td>
<td>−79.21824</td>
<td>AMD (A12)</td>
<td>Treated (T12)</td>
<td>Doser</td>
</tr>
<tr>
<td>Abram Creek headwaters</td>
<td>39.21855</td>
<td>−79.22520</td>
<td>AMD (A13)</td>
<td>Treated (T13)</td>
<td>Doser</td>
</tr>
</tbody>
</table>

NS not sampled, Downstream site is downstream of treatment, LS sand limestone sand, – no treatment, Doser Lime doser
we used a combination of multivariate statistics and ordination procedures. Prior to analyses, all water chemistry variables except pH were log transformed to approximate assumptions of parametric statistics. Alkalinity was normalized after adding 1 mg/L CaCO₃ equivalents to its value, and total acidity was removed from the analysis due to its strong dependence on other chemical elements (Merovich et al. 2007). Cd was not included in the analysis because all concentrations returned by the lab were below the detection limit. First, we used principal component analysis (PCA) to analyze the water chemistry data pre- and post-treatment. Principal components (PCs) with eigenvalues >1.0 were considered significant (McGarigal et al. 2000). Water chemistry parameters were considered strongly correlated to a PC if their factor loadings had an absolute value >0.5 (McGarigal et al. 2000). Separation of water samples (e.g., AMD-type) in ordination space was used to interpret the degree of difference in water quality (i.e., chemical make-up) relative to other sample types (e.g., treated-type). We used multivariate analysis of variance (MANOVA) to determine if a priori stream types had statistically different water chemistry signatures. Analysis of variance (ANOVA) and Tukey’s HSD post-tests were used to determine which specific water chemistry parameters were statistically different among stream types.

To further examine the chemical response to treatment, we calculated acid loads and the net acidity at the mouth of the watershed and major tributaries pre- and post-treatment. Acid and alkalinity loads were calculated by multiplying flow in volume per minute by acidity or alkalinity concentration in CaCO₃ equivalents, respectively. Final values were converted to tons/year. Flows for each of the major tributaries receiving AMD treatment were calculated by using the flow with drainage area ratio method (Emerson et al. 2005). To calculate drainage area, we used a hydrologic model in ArcMap Version 10.1 (Environmental Systems Research Institute, Redlands, CA, USA) to find the total flow accumulation values for each major tributary receiving AMD treatment. Net acidity was calculated by subtracting alkalinity from calculated acidity. Additionally, we used MANOVA to determine if RVHA parameters differed between a priori stream types.

To investigate the response of fish and macroinvertebrate communities to watershed-based remediation, we used nonmetric multidimensional scaling (NMDS; Bray-Curtis distance coefficient). To interpret the gradient structure in the NMDS solutions at acceptable stress levels, we labeled samples (sites) in ordination space with a priori stream types and added to the ordination the weighted mean positions of selected taxa. We also correlated biological metrics and water chemistry parameters to the ordinations. Correlations were considered statistically significant when \( p < 0.05 \) (for 999 permutations of the data). To statistically test whether differences existed among communities from different stream types as suggested by NMDS ordinations, we used analysis of variance using distance matrices (ADONIS; i.e., nonparametric multivariate analysis of variance), with statistical significance \( (p < 0.05) \) evaluated with 999 permutations. In addition, the West Virginia Stream Condition Index (WVSCI), a benthic macroinvertebrate family-level index of biotic integrity (IBI), and the Genus Level Index of Most Probable Stream Status (GLIMPSS), a genus-level benthic macroinvertebrate IBI, were used to quantify ecological integrity at each sample site (Gerritsen et al. 2000; Pond et al. 2008). We calculated five benthic macroinvertebrate community metrics. These included Ephemeroptera, Plecoptera, Trichoptera (EPT) genus richness, genus richness, % EPT families, % Chironomidae, and % 2 dominant for each site (see Pond et al. 2008). For fish, we calculated seven community metrics. These included species richness, number of benthic species, sensitive species richness, proportion of tolerant individuals, proportion of invertivore-piscivore individuals, proportion of macro-omnivores, and proportion of gravel spawning species for each site (see McCormick et al. 2001). ANOVA and Tukey’s HSD post-tests were used to determine which specific benthic macroinvertebrate and fish metrics were statistically different among stream types. Paired \( t \) tests were used to compare macroinvertebrate IBIs’ pre- and post-treatment. All statistical analyses were conducted in the R statistical environment Version 3.0.2 (R Development Core Team 2013). NMDS and ADONIS were performed with the package vegan (Oksanen et al. 2013).

To address our second objective, we calculated ecological units (EUs; sensu Petty and Thorne 2005; Merovich and Petty 2007) to quantify ecological function of stream segments in the Abram Creek watershed. EUs are units of stream length (km) weighted by an ecological function that reflects the quality of the stream segment (Petty and Thorne 2005). For example, a stream segment with a length of 2.5 km and a weighting value of 0.70 would possess an EU value of 1.75 km. The highest quality stream segments...
receive weighting values of 1.0, implying that these segments are functioning at 100% of that expected for stream segments in that region. EUs therefore represent ecological value in units of river length and can be viewed as the availability of ecologically functioning stream habitat (Petty and Thorne 2005).

Using this concept, we developed a way to first estimate conditions before impairment, which we termed Historic EUs (HEUs). HEUs were calculated as stream segment length (SL) × ecological potential (EP; see below). We then quantified lost EUs (LEUs; HEUs − Pre-EUs) relative to current conditions before treatment (Pre-treatment EUs; Pre-EUs). Pre-EUs were calculated as SL × EP × ecological condition before treatment (EC; see below). LEUs were used to estimate recoverable EUs (REUs) as a result of remediation efforts. REUs were calculated as LEUs × ecological remediability (ER; see below). To predict future EUs (FEUs), we then added back REUs to Pre-EUs (Pre-EUs + REUs). Next, we calculated EUs 3 years after treatment (Post-treatment EUs; Post-EUs) in the same way as Pre-EUs but with EC values after treatment. Finally, we compared Post-EUs to FEUs for each sampling location. Table 2 details how these units were calculated using three terms that varied by weighting functions: EP, EC, and ER (Fig. 2). Further, we developed these pre- and post-treatment EUs as well as the expected REUs and FEUs (Table 2) for four different aquatic life endpoints as discussed below: brook trout fishery EUs, stocked trout fishery EUs, macroinvertebrate diversity EUs, and overall fishery EUs, which combined elements of brook trout and stock trout fishery EUs (sensu Petty et al. 2008).

First, the EP function explains the prospective value of a stream reach as ecological habitat given successful remediation and is based on drainage area (DA). For example, the ecological potential function for the brook trout fishery EU is set to 1.0 for DA 0–40 km², then declines linearly for DA >40 km² (Fig. 2a), because native brook trout ecology is concentrated within small drainage areas (Petty and Thorne 2005). Conversely, the ecological potential function for the stocked trout fishery EU is set to 0.0 for DA 0–20 km², then increases linearly for DA >20 km² (Fig. 2a), because stocked trout fisheries are typical of larger drainages where recreational use potential is high (Heidinger 1999). The ecological potential function for the macroinvertebrate diversity EU, which is based on WVSCI (see below), is set to 1.0 for all drainage areas because streams of all sizes should be functioning at full potential in terms of this invertebrate IBI (Fig. 2a).

On the other hand, the EC function explains the current value of the stream reach as habitat for biological diversity or as a fishery and is predicted based on WVSCI scores. WVSCI ranges from 0 to 100 with higher scores reflecting better stream ecosystem condition (Gerritsen et al. 2000). For example, the EC function for the macroinvertebrate diversity EU is 0.0 for WVSCI 0–40, because conditions are very poor in that range (Gerritsen et al. 2000). The function then increases linearly to 1.0 for WVSCI 40–80 as conditions improve (Fig. 2b). The function does not increase to 1.0 for the brook trout fishery EU until WVSCI reaches 80.0 (Fig. 2b), because brook trout are more sensitive to overall ecological condition (Petty and Thorne 2005). The function increases to 1.0 for the stocked trout fishery EU when WVSCI reaches 60.0 (Fig. 2b), because stocked trout are generally more tolerant of sub-optimal conditions of a put-and-take fishery (Trushenski et al. 2010).

Finally, the ER function relates the likelihood of remediation to drainage area and treatment type. In this study, at-source treatment refers to passive AMD remediation systems near the source of AMD, while in-stream approaches refer to liming dosers and limestone
sand dumpsites further away from AMD sources. For example, the ER weighting function increases linearly to 1.0 as DA increases to 20 km² for at-source treatment, whereas for in-stream treatment, DA must reach 10 km² before remediability begins to accrue weight (Fig. 2c). Remediation of small streams (less than 2 km²) is unlikely for both at-source and in-stream treatment approaches because the proximity to chemical treatment creates a chemical mixing zone that prevents biological recovery (McClurg et al. 2007). Furthermore,
remediation is expected to be less likely using in-stream approaches as compared to at-source approaches, especially in small- to moderate-size streams (Petty et al. 2008). Therefore, low remediability is associated with small drainage area within close proximity to treatment. Remediability increases as stream size and distance to treatment increases (McClurg et al. 2007).

To spatially analyze pre- and post-treatment ecological conditions, we assigned each of our observed pre- and post-treatment WVSCI scores to their appropriate segment-level watershed (Merovich and Petty 2007; Strager et al. 2009). Next, we linearly interpolated WVSCI scores between segment-level watersheds bounded by observed scores (Merovich and Petty 2007). This was necessary to assign WVSCI scores to un-sampled segment-level watersheds throughout the stream continuum. Finally, we accumulated the segment-level watershed values for each EU type from the headwaters to the mouth to obtain an ecological currency in stream kilometers at the watershed scale. We ran two-sample Kolmogorov-Smirnov tests to determine if statistical differences existed between the cumulative distribution curves of observed post-treatment EUs versus expected (i.e., future) post-treatment EUs for each EU type in the watershed.

Results

Physicochemical

We observed high variability in water chemistry throughout the watershed. PCA revealed four important dimensions of variation (eigenvalues >1.0). Only PC 1 and PC 2 were interpreted, because they were statistically different among stream types (Table 3). Combined, PC 1 and PC 2 explained 69% of the variance observed in water chemistry. PC 1 is interpreted as a pollution gradient explaining 46% of the variance. PC 1 was strongly correlated with Al, Ba, Cu, Co, Cr, Ni, Zn, SO4$^{2-}$, Fe, Mg, and Mn concentrations in the positive direction (Fig. 3). Alkalinity and pH were correlated negatively with PC 1 (Fig. 3). PC 2 is interpreted as a hardness gradient explaining 23% of the variance in water chemistry data. PC 2 was strongly correlated with specific conductance, Ca, Mg, Na, and SO4$^{2-}$ in the positive direction (Fig. 3). Cu and Cr were correlated negatively with PC 2 (Fig. 3). In terms of water chemistry, MANOVA showed a statistical difference between stream types ($F = 16.923; df = 2, 29; p = 5.6 \times 10^{-11}$). ANOVA ($p < 0.05$) showed Ni, Zn, and Mn were statistically different among all stream types (Table 3). Al, Ba, Cu, Co, Cr, and Fe concentrations were statistically higher in AMD stream types compared to treated and reference stream types, while pH and alkalinity concentrations were statistically higher in treated and reference stream types compared to AMD stream types (Table 3). Specific conductance, SO4$^{2-}$, Ca, and Mg concentrations were statistically higher in AMD and treated stream types compared to reference stream types, and Cl$^-$, Na, and Se concentrations were all statistically equivalent among stream types (Table 3).

Acid loads and net acidity calculated at the mouth of the watershed and all major tributaries receiving AMD treatment declined, except at Laurel Run. Specifically, the acid load at the mouth of Abram Creek was reduced from 330 to 34 t/year. Emory Creek was observed to have the greatest reduction in net acidity compared to other treated tributaries at ~220 t/year. Laurel Run was observed to gain approximately 16 t/year of acidity (Table 4). We observed no statistical difference in RVHA scores among stream types (MANOVA $F = 1.0648; df = 2, 30; p = 0.4161$).

Benthic macroinvertebrates and fish

Biological conditions varied widely throughout the watershed. NMDS ordinations indicate that invertebrate and fish assemblages are responding positively to water chemistry improvements (Figs. 4 and 5). For macroinvertebrates, percent Ephemeroptera increased strongly at treated sites (Fig. 4c). Taxa predominantly correlated to improvements were from the family Baetidae, specifically genera *Planditus* and *Acentrella* (Fig. 4d). The global ADONIS revealed differences in both macroinvertebrate ($F = 2.4558; p = 0.004; R^2 = 0.14$) and fish ($F = 6.0756; p = 0.001; R^2 = 0.38$) assemblages from different stream types. Subsequent pairwise comparisons found that macroinvertebrate assemblages from AMD streams were statistically different from treated streams ($F = 2.4063; p = 0.033; R^2 = 0.09$) and reference streams ($F = 3.0205; p = 0.013; R^2 = 0.15$), but treated streams were not statistically different compared to reference streams ($F = 1.8881; p = 0.07; R^2 = 0.11$). ADONIS pair-wise comparisons for fish communities revealed that fish communities from AMD streams were statistically different from treated streams ($F = 4.1008; p = 0.005; R^2 = 0.24$) and reference streams ($F = 5.9776; p = 0.004; R^2 = 0.40$), and treated streams were statistically different compared to reference streams ($F = 7.758; p = 0.001; R^2 = 0.30$).
Paired *t* tests showed that both WVSCI (*p* = 0.0001, *df* = 12) and GLIMPSS (*p* = 0.0002, *df* = 12) increased statistically post-treatment. ANOVA found that GLIMPSS and the EPT genus richness metric were statistically different among all stream types (Table 5). The % Chironomid metric was statistically higher in AMD streams compared to treated and reference streams, while % EPT families and WVSCI were statistically higher in treated and reference streams compared to AMD streams (Table 5).

We found fish at nine stream segments that previously had none, two of which had brook trout (*Salvelinus fontinalis*; 11 at Emory Creek and 9 at Glade Run) and one of which had smallmouth bass (*Micropterus dolomieu*; 26 at Abram Creek upstream of Emory Creek). All sites that previously lacked fish had creek chub (*Semotilus atromaculatus*). ANOVA tests found that fish species richness was statistically higher in treated and reference streams compared to AMD streams, while the proportion of macro-omnivores and gravel spawning species were statistically different among all stream types. Also, the proportion of invertivore-piscivore individuals was statistically higher in treated streams compared to AMD and reference streams (Table 5).

Ecological units

We estimated that the Abram Creek watershed historically possessed 55.6 km of overall fishery EUs (Table 6 and Fig. 6a), 55.8 km of macroinvertebrate diversity EUs (Table 6 and Fig. 6b), 49.4 km of brook trout fishery EUs

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**Table 3** Means and standard deviations (SD) of water chemistry parameters and principal component (PC) 1, 2, 3, and 4 scores for each stream type

<table>
<thead>
<tr>
<th>Stream type</th>
<th>AMD (n = 14)</th>
<th></th>
<th>Treated (n = 13)</th>
<th></th>
<th>Reference (n = 4)</th>
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<tr>
<td></td>
<td>Mean</td>
<td>SD</td>
<td>Mean</td>
<td>SD</td>
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<td>Specific conductance</td>
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<td>69.59</td>
<td>386.85</td>
<td>90.40</td>
<td>118.8</td>
<td>68.69</td>
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<td>Alkalinity</td>
<td>4.03</td>
<td>3.39</td>
<td>15.31</td>
<td>7.84</td>
<td>24.57</td>
<td>13.75</td>
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<tr>
<td>Al</td>
<td>0.47</td>
<td>0.82</td>
<td>0.03</td>
<td>0.04</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>Ba</td>
<td>0.07</td>
<td>0.02</td>
<td>0.03</td>
<td>0.00</td>
<td>0.04</td>
<td>0.01</td>
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<tr>
<td>Cu</td>
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<td>0.00</td>
<td>0.01</td>
<td>0.00</td>
<td>0.01</td>
<td>0.00</td>
</tr>
<tr>
<td>Cl−</td>
<td>4.35</td>
<td>2.32</td>
<td>5.05</td>
<td>2.82</td>
<td>6.45</td>
<td>4.89</td>
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<tr>
<td>Co</td>
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<td>0.04</td>
<td>0.01</td>
<td>0.01</td>
<td>0.01</td>
<td>0.00</td>
</tr>
<tr>
<td>Cr</td>
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<td>0.01</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>Na</td>
<td>3.88</td>
<td>1.88</td>
<td>5.72</td>
<td>1.63</td>
<td>3.86</td>
<td>2.76</td>
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<tr>
<td>Ni</td>
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<td>0.03</td>
<td>0.03</td>
<td>0.01</td>
<td>0.01</td>
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</tr>
<tr>
<td>Se</td>
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<td>0.05</td>
<td>0.03</td>
<td>0.00</td>
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<td>0.01</td>
</tr>
<tr>
<td>Zn</td>
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<td>0.03</td>
<td>0.03</td>
<td>0.01</td>
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<tr>
<td>SO₄²⁻</td>
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<td>136.61</td>
<td>61.13</td>
<td>11.48</td>
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<td>Mg</td>
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<td>5.39</td>
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<tr>
<td>Mn</td>
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<td>PC1</td>
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<td>1.21</td>
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<tr>
<td>PC2</td>
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<tr>
<td>PC3</td>
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<tr>
<td>PC4</td>
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<td>1.27</td>
<td>-0.20</td>
<td>0.88</td>
<td>-0.13</td>
<td>0.69</td>
</tr>
</tbody>
</table>

Means with different letters among stream types are statistically different (*p* < 0.05; analysis of variance, Tukey post-test). Means are reported in milligrams per liter. Specific conductance is reported in microsiemens per centimeter, and alkalinity is reported in milligrams per liter CaCO₃ equivalents.
20.7 km of stocked trout fishery EUs (Table 6 and Fig. 6d) throughout the study area. As a result of AMD impacts, pre-treatment EUs indicate that only 30% of the historic overall fishery value, 42% of the macroinvertebrate diversity, 28% of the brook trout, and 14% of the stock trout fishery remained in 2008 (Table 6 and Fig. 6). The watershed-based remediation plan developed and implemented was expected to recover 66% of the historic overall fishery value, 67% of the macroinvertebrate diversity, 61% of the brook trout, and 92% of the stock trout fishery (Table 6 and Fig. 6). The observed post-treatment EUs indicate that 52% of the historic overall fishery value, 74% of the macroinvertebrate diversity, 33% of the brook trout, and 72% of the stocked trout fishery (Table 6 and Fig. 6) have been recovered 3 years after remediation efforts were implemented.

The two-sample Kolmogorov-Smirnov tests suggest that the observed post-treatment macroinvertebrate diversity, stocked trout fishery, and overall fishery EUs accumulated at a statistically equivalent rate along the river compared to the predicted post-treatment macroinvertebrate diversity, stocked trout fishery, and overall fishery EUs ($p = 0.6465$, 0.1717, and 0.0966, respectively). However, the observed post-treatment brook

<table>
<thead>
<tr>
<th>Site</th>
<th>2008</th>
<th>2013</th>
<th>Δ net acidity</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Acid load</td>
<td>Alkalinity load</td>
<td>Net acidity</td>
</tr>
<tr>
<td>Abram Creek at mouth</td>
<td>228.0</td>
<td>330.3</td>
<td>−102.3</td>
</tr>
<tr>
<td>Emory Creek</td>
<td>130.4</td>
<td>30.2</td>
<td>100.2</td>
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<tr>
<td>Glade Run</td>
<td>13.6</td>
<td>11.4</td>
<td>2.2</td>
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<tr>
<td>Laurel Run</td>
<td>41.4</td>
<td>36.0</td>
<td>5.4</td>
</tr>
<tr>
<td>Little Creek</td>
<td>56.8</td>
<td>3.6</td>
<td>53.3</td>
</tr>
<tr>
<td>Abram Creek headwaters</td>
<td>25.6</td>
<td>0</td>
<td>25.6</td>
</tr>
</tbody>
</table>

Negative values of Δ (delta) net acidity indicate decline
trout fishery EUs accumulated at a statistically lower rate compared to the predicted post-treatment brook trout fishery EUs ($p = 0.0009$) (Table 6 and Fig. 6).

Discussion

In this study, we evaluated expected outcomes of a watershedscale remediation initiative. We documented notable improvements in environmental conditions related to AMD treatment, but biotic responses, especially measures of fishery quality, fell short of meeting expectations, despite evidence of reconnection of the stream network. Consequently, in this study, we have related improvements in watershed conditions to ecological benefits expected prior to remediation actions.

Our finding of improved water quality as a result of AMD treatment is consistent with numerous studies conducted in this region (Trout Unlimited (TU) 2011, Simon et al. 2012). We characterized the overall chemical
response to AMD treatment as a transition from metal-laden acidic waters to hard waters with elevated specific conductance and SO$_4^{2-}$ concentrations. Although this new water chemistry signature is different from reference conditions, biological assemblages have improved with these chemical improvements.

In our study, the genera predominantly responsible for improvements in macroinvertebrate assemblages at treated sites were from the family Baetidae, specifically the genera *Plauditus* and *Acentrella*. An explanation of the improved but altered assemblages is that treated streams were being colonized by pioneer invertebrate assemblages. Vieira et al. (2004) characterized Baetid mayflies as being strong larval dispersers and found a Baetid species to rapidly recolonize and dominate benthic communities during early post-disturbance years.

Fig. 5 Nonmetric multidimensional scaling (NMDS) ordination of fish samples (Bray-Curtis distance coefficient) in two dimensions showing sites labeled by stream type (a), water chemistry vectors (b), fish metrics (c), and weighted mean positions of selected species (d). Stress = 0.11 in the two-dimensional solution. Stream type abbreviations as in Fig. 3. Cond specific conductance, BenthicSpp no. of benthic species, ProTol proportion tolerant, SRICH species-level richness, SenSRICH sensitive species richness, ProL.P proportion of invertivore-piscivores, ProM.O proportion of macro-omnivores, ProGSpawn proportion of gravel spawning species, SEAT Semotilus atromaculatus (creek chub), SAFO Salvelinus fontinalis (brook trout), ETFL Etheostoma flabellare (fantail darter), COCA Cottus caeruleommentum (blue ridge sculpin), RHAT Rhinichthys atratulus (eastern blacknose dace), CACO Catostomus commersonii (white sucker), LECY Lepomis cyanellus (green sunfish), MIDO Micropterus dolomieu (smallmouth bass), CAAN Campostoma anomalum (stoneroller), AMRU Ambloplites rupestris (rock bass). ADONIS $p$ value = 0.001
Additional studies examining successional sequences have found the family Baetidae to be early colonists after catastrophic watershed scale disturbances (Flory and Milner 2000, Zuellig et al. 2002). Strong dispersal capability coupled with low habitat specificity defines these taxa as pioneer species (Gore 1982). Thus, it is evident that benthic macroinvertebrate assemblages are improving given the improved chemical environment we have characterized in the Abram Creek watershed.

Fish assemblages have also improved. However, fish assemblages sampled from treated reaches were quite different compared to reference reaches. Despite differences in community structure among stream types, we found fish present at nine assessment sites which previously had none. All sites that previously lacked fish had creek chub, the only species of fish found in upper Abram Creek; however, further downstream from the three lime dosers, we observed brook trout, smallmouth bass, and eastern blacknose dace (*Rhinichthys atratulus*). Brook trout were found at two sites which previously had none. In 2007, before project completion, the Johnnycake Run tributary network was the only intact fishery remaining in the Abram Creek drainage. This movement of a sensitive species into stream

### Table 5

<table>
<thead>
<tr>
<th>Water quality type</th>
<th>AMD (n = 14)</th>
<th>Treated (n = 13)</th>
<th>Reference (n = 4)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean SD</td>
<td>Mean SD</td>
<td>Mean SD</td>
</tr>
<tr>
<td>WVSCI</td>
<td>47.75 a 17.64</td>
<td>67.39 b 13.76</td>
<td>85.17 b 1.95</td>
</tr>
<tr>
<td>GLIMPSS</td>
<td>26.03 a 15.07</td>
<td>41.50 b 14.18</td>
<td>66.70 c 6.82</td>
</tr>
<tr>
<td>EPT genus richness</td>
<td>5.36 a 4.75</td>
<td>9.85 b 4.56</td>
<td>16.80 c 3.96</td>
</tr>
<tr>
<td>Genus richness</td>
<td>13.00 a 7.68</td>
<td>19.23 b 7.70</td>
<td>27.40 b 1.52</td>
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<tr>
<td>% EPT families</td>
<td>28.87 a 25.28</td>
<td>59.13 b 23.00</td>
<td>61.75 b 8.94</td>
</tr>
<tr>
<td>% Chironomidae</td>
<td>45.68 a 18.44</td>
<td>25.95 b 17.65</td>
<td>21.11 b 4.39</td>
</tr>
<tr>
<td>% 2 dominant</td>
<td>66.24 a 12.78</td>
<td>61.20 a 12.08</td>
<td>44.40 b 11.13</td>
</tr>
<tr>
<td>Fish species richness</td>
<td>0.73 a 1.75</td>
<td>2.77 b 1.96</td>
<td>3.75 b 2.43</td>
</tr>
<tr>
<td># of benthic species</td>
<td>0.00 a 0.00</td>
<td>0.08 a 0.28</td>
<td>0.50 b 0.53</td>
</tr>
<tr>
<td>Sensitive species richness</td>
<td>0.13 a 0.35</td>
<td>0.23 a 0.44</td>
<td>0.75 b 0.46</td>
</tr>
<tr>
<td>Proportion of tolerant individuals</td>
<td>0.17 a 0.36</td>
<td>0.86 b 0.28</td>
<td>0.78 b 0.22</td>
</tr>
<tr>
<td>Proportion of I-P individuals</td>
<td>0.15 a 0.33</td>
<td>0.61 b 0.3</td>
<td>0.13 a 0.12</td>
</tr>
<tr>
<td>Proportion of macro-omnivores</td>
<td>0.05 a 0.15</td>
<td>0.31 b 0.28</td>
<td>0.71 c 0.25</td>
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<tr>
<td>Proportion of gravel spawning species</td>
<td>0.05 a 0.13</td>
<td>0.32 b 0.30</td>
<td>0.76 c 0.26</td>
</tr>
</tbody>
</table>

Means with different letters among stream types are statistically different ($p < 0.05$; analysis of variance, Tukey post-test)


### Table 6

<table>
<thead>
<tr>
<th>Aquatic life endpoints</th>
<th>HEUs</th>
<th>Pre-EUs</th>
<th>LEUs</th>
<th>REUs</th>
<th>FEUs</th>
<th>Post-EUs</th>
<th>K-S Test p values</th>
</tr>
</thead>
<tbody>
<tr>
<td>Macroinvertebrate diversity EU</td>
<td>55.8</td>
<td>23.3 (42%)</td>
<td>32.5 (58%)</td>
<td>14.0 (25%)</td>
<td>37.3 (67%)</td>
<td>41.1 (74%)</td>
<td>0.65</td>
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<tr>
<td>Overall fishery EU</td>
<td>55.6</td>
<td>16.7 (30%)</td>
<td>38.9 (70%)</td>
<td>19.8 (36%)</td>
<td>36.5 (66%)</td>
<td>28.9 (52%)</td>
<td>0.10</td>
</tr>
<tr>
<td>Brook trout fishery EU</td>
<td>49.4</td>
<td>13.8 (28%)</td>
<td>35.6 (72%)</td>
<td>16.4 (33%)</td>
<td>30.2 (61%)</td>
<td>16.2 (33%)</td>
<td>0.001</td>
</tr>
<tr>
<td>Stocked trout fishery EU</td>
<td>20.7</td>
<td>2.9 (14%)</td>
<td>17.8 (86%)</td>
<td>16.2 (78%)</td>
<td>19.1 (92%)</td>
<td>14.8 (72%)</td>
<td>0.17</td>
</tr>
</tbody>
</table>

Values in parentheses indicate the percent of HEUs

*HEUs* historic EUs, *Pre-EUs* pre-treatment EUs, *LEUs* lost EUs, *REUs* recoverable EUs, *FEUs* future EUs (i.e., predicted), *Post-EUs* post-treatment EUs, *K-S test* two-sample Kolmogorov-Smirnov test
segments which previously lacked fish suggests that previously isolated, poor-quality streams were improved and reconnected to distal sub-watersheds via improvements to the mainstem. McClurg et al. (2007) found that limestone treatment did not fully recover total fish biomass to reference conditions in acid-impacted watersheds of the Allegheny Plateau. They accredited this shortcoming to the isolation of treated streams in a network of acidic watersheds. Consequently, stream remediation plans should focus on recovering stream ecosystems as connected networks rather than isolated reaches (McClurg et al. 2007).

Despite obvious improvements in water quality and ecological communities, we did not observe watershed scale recovery as expected. While the macroinvertebrate diversity EUs exceeded expectations, the brook trout, stocked trout, and overall fishery EUs did not meet our expected predictions. Several details may explain why the observed conditions fell short of our predicted outcomes for fish communities. First, AMD treatment did not decrease SO$_4^{2-}$ concentrations or specific conductance to levels comparable to reference conditions. Sulfate concentrations are not expected to decrease with the treatment technologies used, and high levels of specific conductance are common for treated mine water (DeNicola and Stapleton 2016). Pond et al. (2008) found biological impairment when specific conductance was greater than 500 $\mu$S/cm. A more recent study observed biological impairment when specific conductance reached 250 $\mu$S/cm (Merriam et al. 2011). Freund and Petty (2007) observed a negative correlation between SO$_4^{2-}$ concentration and biotic indices. These results suggest that elevated specific conductance and SO$_4^{2-}$ concentrations may restrict biological communities from recovery. Future remediation plans should focus efforts on the reduction of SO$_4^{2-}$ and specific conductance concentrations.
Secondly, treatment at Laurel Run was not effective and net acidity increased, probably because of current mining in the sub-watershed. A mining company holds a refuse disposal permit, and AMD seeps above the stream are affecting the limestone sand treatment (J. Baczuk; West Virginia Department of Environmental Protection, personal communication). Laurel Run was also the only site receiving AMD treatment not to have fish despite a minimal increase in benthic macroinvertebrate IBI scores. Several other studies have observed little to no recovery and delayed recovery of invertebrate communities following remediation as well (Gunn et al. 2010; Louhi et al. 2011; DeNicola and Stapleton 2016). These observations are often attributed to varying levels of AMD inputs and inconsistent treatment conditions, episodic acidification, and scarcity of organic matter input (Bradley and Ormerod 2002; Simmons et al. 2005; Gunn et al. 2010; DeNicola and Stapleton 2016). Thus, current AMD treatment may not be enough to mitigate active mining activities in this case; therefore, we recommend that additional treatment in Laurel Run be implemented as an important next step in the adaptive watershed management cycle.

Another limiting factor was the exceptional length of the chemical mixing zone. In general, we found that observed post-treatment EUs accumulated downstream of treatment at a much slower rate than predicted post-treatment EUs (Fig. 6), with the exception of the cumulative macroinvertebrate diversity EU (Fig. 6b). For example, our remediability functions predicted that overall fishery EUs would start accruing after river kilometer 7.4, at the point where cumulative DA becomes capable of supporting a fishery (Fig. 6a). Post-treatment analysis indicates that overall fishery EUs do not substantially start accruing until river kilometer 17.2, at the point where WVSCI scores became capable of supporting a fishery (Fig. 6a). McClurg et al. (2007) found evidence that a sacrificial zone of ecological impairment existed 2–3 km from limestone sand treatment in acid deposition-impacted watersheds of WV. Our finding of brook trout in Emory Creek approximately 2.5 km downstream from the limestone sand is consistent with this impairment threshold. Water chemistry data from Emory Creek is characteristic of acidic conditions caused by acid deposition; there are no AMLs or surface mines upstream. Acid precipitation severely impairs mountain streams in this region and is distinguished by depressed pH, very low specific conductance, and elevated concentrations of dissolved Al in surface waters (Merovich et al. 2007; Driscoll et al. 2011). The limestone sand applied to Emory Creek is neutralizing the acid deposition from the left fork of the Emory Creek headwaters helping to keep the acidity lower at the mouth of the Emory Creek drainage. However, sacrificial zones associated with AMD treatment are much greater because dissolved metal concentrations and overall water quality tend to be worse than in watersheds impacted solely by acid precipitation (Merovich et al. 2007). Consequently, the type of pollution being treated (i.e., acid deposition or AMD) and distance to the type of treatment should be factored into remediability weighted functions that calculate recoverable EUs under a given remediation alternative.

Finally, another limiting factor is the regional context in which Abram Creek watershed is located. Variability in metacommunity dynamics (i.e., species sorting and mass effects) probably influenced our ability to predict biological recovery at the watershed scale (Merriam et al. 2015). Before treatment began, we estimated that only 30% of the overall fishery value in the watershed remained, and this was isolated to Johnnycake Run, a small sub-watershed surrounded by impaired waters. Improved watershed scale water quality reconnected Johnnycake Run to Abram Creek, but regional conditions in the Potomac may limit the ability of Abram Creek watershed to recover. The North Branch of the Potomac River basin is well known for its history of problems associated with AMLs. While conditions in the North Branch of the Potomac have greatly improved and now support a put-and-take trout fishery, biological conditions remain erratic and sub-optimal (Hansen et al. 2010). Local communities are a product of the regional species pool (Sundermann et al. 2011) and therefore unsupportive biological factors such as the lack of local (alpha) or regional (beta) diversity may restrict fisheries expectations from being realized in a timely manner at the watershed scale in Abram Creek. Restoration should seek to prioritize efforts toward needs within good regions rather than poor regions. A multiscale approach ensures that restoration efforts build on existing high-quality conditions (Merovich et al. 2013). Nevertheless, restoration effort in Abram Creek is a success in terms of improving water quality and local environmental conditions.

The use of EUs in our analysis allowed us to quantify the response of ecological conditions to treatment at both the local and watershed scale. By assigning an ecological currency to individual segment-level watersheds, we were able to document the effect of treatment on the ecological recovery of specific drainages, as well as the cumulative benefits of treatment to the stream network. Our analysis is
the first that we know that provides a means to evaluate remediation outcomes with those expected. While our weighting functions for EU calculation represent hypotheses of watershed processes, they are our best approximation from which to develop expectations. Further work to refine these functions and develop additional needed models would be useful.

There are a number of reasons why having explicit expectations of remediation outcomes is useful. First, in the initial phases of designing treatment scenarios at the watershed scale, quantifying expected benefits aids identification of treatment technologies and locations within the watershed to maximize remediation benefits (Petty et al. 2008). Once the remediation plan is underway, comparisons between current conditions and expected outcomes provide information to direct a posteriori adaptive management actions where there is the greatest need. For example, where watershed scale benefits are not accumulating as predicted, improvement efforts can be directed at those treatment locations responsible for under-performance. Finally, comparing observed performance against expected outcomes among the various aquatic life end points can suggest which watershed management goals (e.g., brook trout vs. stocked trout, etc.) are most feasible given the conditions in the region. A similar process is possible for other end points depending on the regional conditions or watershed objectives (e.g., warmwater fisheries).

Conclusions

We were able to complete the adaptive watershed management cycle for the Abram Creek watershed remediation plan. This post-treatment assessment documented a reconnection of the watershed and significantly improved ecological conditions in Abram Creek. These improvements were notably observed downstream of liming dosers and limestone sand dump sites throughout the watershed, with the exception of Laurel Run which may have been compromised due to ongoing mining activity. Only 3 years passed since project completion and communities that reflect reference conditions may require more time. We demonstrated that it is possible to (1) quantify the degree of ecological loss from mining-related stressors on stream ecosystems, (2) estimate the amount of ecological habitat that can be recovered through remediation programs, and (3) quantify the degree of post-treatment ecological recovery at both the stream segment and watershed scale. We expect this general framework developed for mined watersheds can be translated to other anthropogenic stressors. Finally, we believe that our framework for evaluating expected outcomes will provide an accounting system to identify where adjustments are needed to bring remediation better in line with expectations. Doing so will expedite remediation, rather than letting poor responses unknowingly lag. Continued long-term monitoring of the Abram Creek watershed is needed to further evaluate remediation progress valuable toward advancing the science of watershed remediation ecology.

Acknowledgements

We thank Donna Hartman, Eric Merriam, Eric Miller, Brock Huntsman, and Alison Anderson for their help in field sampling and laboratory analysis. We thank Jim Baczuk from WVDEP Office of Abandoned Mine Lands and Reclamation for sharing his expertise and overlooking the operation and maintenance for the Abram Creek watershed remediation project. This work was funded, in part, by the Appalachian Research Initiative for Environmental Science, the West Virginia Water Research Institute, and West Virginia University, School of Natural Resources. All sampling was approved by The Institutional Animal Care and Use Committee (IACUC) of West Virginia University (most recent protocol number 11–0507).

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