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PRIMARY RESEARCH PAPER

Ecological function of constructed perennial stream channels on reclaimed surface coal mines

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Abstract Mountaintop removal–valley fill mining results in the conversion of steep, forested headwater catchments to low gradient and open canopy channels. We compared the ecological functions of five reference stream channels to five constructed channels (age ranging from 3 to 20 years) on reclaimed mines in southern West Virginia. Variables included stream flow, habitat, water chemistry, riparian vegetation, organic matter (OM) processing, and invertebrate and amphibian communities. Although dissolved metal concentrations remained low, constructed channels produced significantly higher levels of conductivity and total dissolved solids as compared to reference streams. Macroinvertebrate and amphibian richness were comparable between constructed and reference channels; however, there was a distinct shift from sensitive lotic taxa in reference channels to tolerant lentic taxa in constructed channels. Constructed channels also had reduced OM decomposition rates. Nevertheless, constructed channels had significantly higher OM retention than reference channels, and consequently exhibited significantly higher overall OM processing and higher dissolved carbon concentrations. As the time since reclamation increased, we observed slight declines in conductivity and significant increases in total invertebrate richness. Our results provide measures of functional equivalencies between reference and constructed streams, which can serve as a basis for informed permitting and mitigation decisions in mined watersheds.

Keywords Headwater stream functions · Mining reclamation · Mountaintop removal–valley fill mining · Stream restoration

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Introduction

Effectively managing the development of large-scale surface mines in the central Appalachian region may be one of the most pressing environmental issues in the U.S. at this time. A comprehensive Environmental Impact Study demonstrated that mountaintop mining/valley fill (henceforth MTM/VF) can cause alteration of natural headwater functions and ecological services (USEPA, 2005). Headwater wetlands and streams provide valuable habitat for numerous wildlife, invertebrate, and plant species (Balcombe et al., 2005a, b; Finn et al., 2011). Moreover, headwaters provide a complex network of ecological services such as flood mitigation, nutrient and organic matter (OM) cycling, aquifer recharge, improving water quality,

and providing timber and other merchantable products (Meyer & Wallace, 2001; Wipfli et al., 2007). Consequently, there is broad concern that a cumulative loss of headwater functions from mining can cause unacceptable impacts to larger waterbodies downstream (Palmer et al., 2010; Lindberg et al., 2011; Merriam et al., 2011).

In an effort to avoid cumulative impacts to watershed function, the Clean Water Act and related policies require that compensatory stream and wetland mitigation must replace ecological functions that are impacted by development (National Research Council, 2001). However, construction of functional stream channels on mine spoil is extremely difficult and in some settings may be impossible. Precipitation tends to infiltrate quickly into the mine spoil. Consequently, stream channels constructed on reclaimed mines often are dry for much of the year (Fritz et al., 2010). Perennial channels must be constructed along the mine perimeter where the mine spoil intercepts unaltered bedrock. However, these channels necessarily are low gradient and develop structural and riparian characteristics that are more similar to wetlands than to the headwater streams that that they are designed to replace.

Our own research over the last several years has focused on improving techniques for restoring and constructing streams and wetlands in the central Appalachian region (Petty & Thorne, 2005; McClurg et al., 2007; Merovich & Petty, 2007; Veselka et al., 2010; Gingerich & Anderson, 2011). Through this research, we have developed methods for maximizing benefits of stream restoration through quantification of ecological currencies and strategic application of restoration technologies at a watershed scale (Poplar-Jeffers et al., 2009). Despite a good understanding of how natural headwater streams and wetlands function, we have not effectively quantified the specific functions that are lost during large-scale surface mine development. Furthermore, we have not quantified the extent to which the functional values of constructed stream channels on reclaimed mines compare to values characteristic of natural headwater systems (but see Fritz et al., 2010). To our knowledge, there are no comprehensive studies of the ecological values of perennial channels constructed on reclaimed surface mines in the Appalachian region. Consequently, our objectives were to: (1) quantify the functional value of perennial stream channels constructed on large surface mines; (2) compare the ecological values of constructed channels to those of intact headwater streams; (3) determine if the functional value of constructed channels improves with mine age; and (4) develop ecological equivalencies that can be used to relate the value of constructed channels to intact headwater ecosystems. Ecological functions were measured in terms of physical and riparian habitats, water quality, biological diversity and composition, and OM retention and decomposition.

Methods

Study area

We studied five perennial stream channels constructed on large surface mines (henceforth "constructed channels") located in the coal-rich region of southwestern West Virginia (Fig. 1). Within the region, typical constructed channels on reclaimed mines are low gradient and occur along the mine perimeter adjacent to valley fills (Fig. 2). The sites chosen for study varied in drainage area and age since reclamation (Table 1) but all maintained perennial flows. Reference sites consisted of five first order perennial streams that represented the best available sites within a reasonable proximity to constructed channel sites (Figs. 1, 2). Reference sites were selected to be unaffected by surface mining or other known stressors. All study reaches were 30× the mean channel width in length with a minimum length of 150 m and a maximum length of 300 m.

Field sampling and laboratory analysis

Habitat quality assessments were performed in June 2008 using classification systems that included: Virginia Unified Stream Method (USACOE, 2007), West Virginia Functional Channel Unit Assessment (USACOE & VADEQ, 2007), Wildland Hydrology's Bank Erosion Hazard Index (Rosgen, 2001), Environmental Protection Agency Rapid Bioassessment Protocol (Barbour et al., 1999), and Ohio Rapid Assessment Method. Vegetation was sampled according to protocols adapted from Balcombe et al. (2005a, b).

Sites were visited seasonally (four visits) from February 2008 to May 2009 to obtain instantaneous measurements of pH, conductivity, dissolved oxygen,

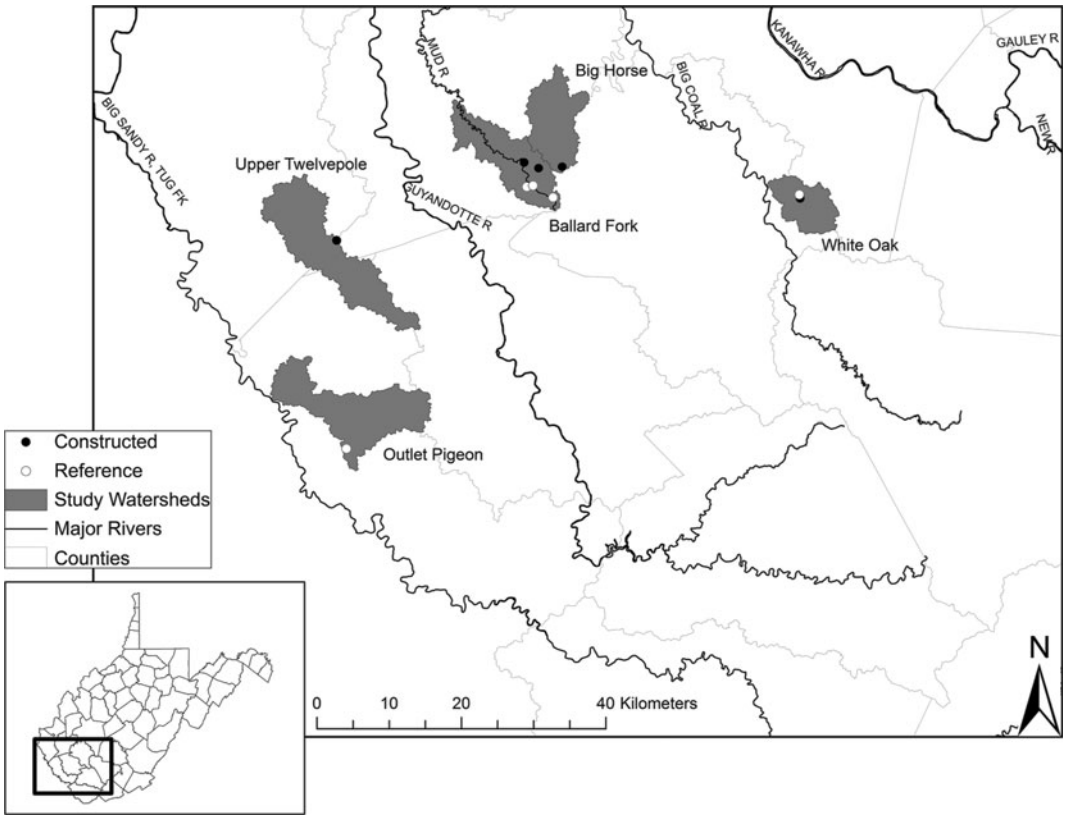

Fig. 1 Study area and site locations for reference streams (*open circles*) and constructed channels (*closed circles*)

and temperature with a Yellow Springs Instrument (YSI) 650 multi-parameter probe equipped with a 600XL sonde. Whole and filtered water samples were taken for analysis of dominant ions, metals, nutrients, and dissolved carbon following protocols by Merovich et al. (2007). Dissolved carbon was measured with a Sievers 5310c laboratory TOC Analyzer. All other chemical analyses were performed by the National Research Center for Coal and Energy at West Virginia University, Morgantown, WV. Alkalinity was measured in $CaCO_3$ equivalents, and conductivity is presented as µS/cm. All other water quality constituents are presented as mg/l. If samples were measured below the method detection limit (MDL), one half of the MDL was used in subsequent analyses. Discharge was calculated from width, depth, and flow measurements taken at each site visit. Flow was measured using a Flow Mate 2000 flow meter. Hourly temperature readings were taken from June 2008 until June 2009 using HOBO U22 Water Temp Pro v2 loggers (Onset Computer Corporation, Pocasset, MA).

Amphibian assemblages were sampled seasonally in early and late spring and early and late summer (March, May, June, and July) of 2008. Adult assemblages were estimated using visual encounter surveys (VES) (Crump & Scott, 1994). Larval surveys were composed of meter-long sweeps with a D-frame net in open water and meter by half-meter area searches in stream channels (Shaffer et al., 1994). Each search was done at 10 random locations over the length of the site.

Macroinvertebrate assemblages were sampled within lotic and lentic habitats in spring and fall 2008 following modified USEPA rapid bioassessment procedures (Balcombe et al., 2005a; Merovich & Petty, 2007). Macroinvertebrate data, however, were analyzed from spring only due to extremely low flow conditions in reference channels during the fall 2008 sample. Within lotic habitats, a D-frame net was used to obtain kick samples (net dimensions 335 × 508 mm with 500 µm mesh) from four targeted riffles at each site. Within lentic habitats, a D-frame net was used to take jab samples at 10 random

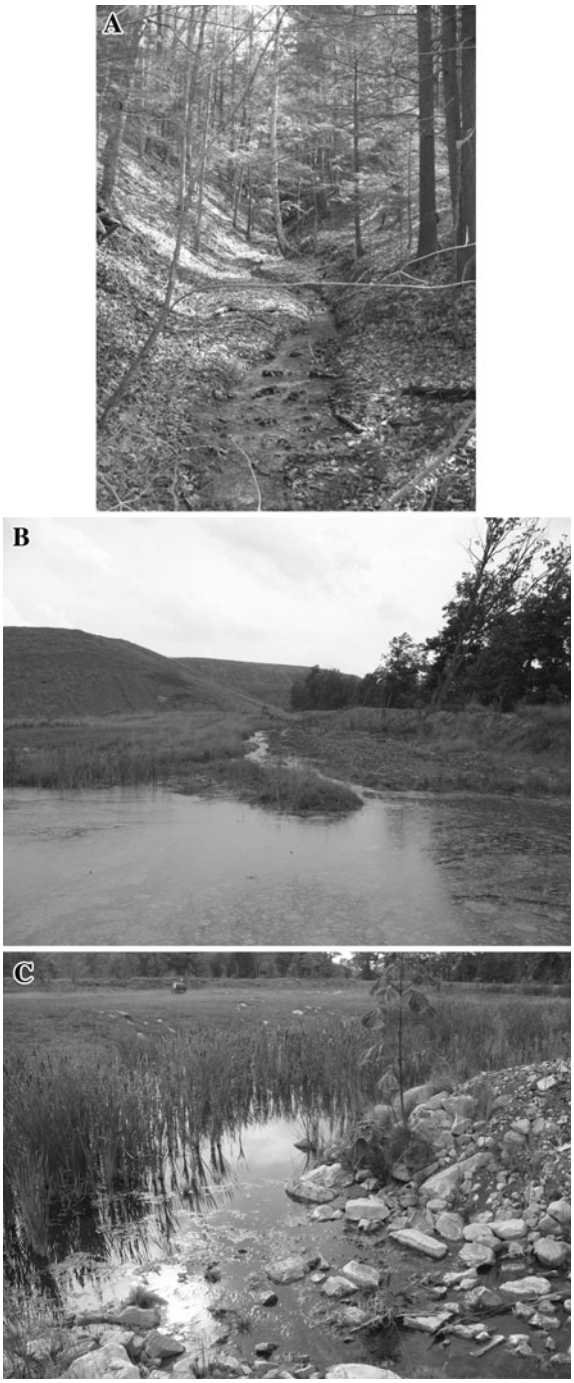

Fig. 2 **A** Typical reference headwater catchment; **B** typical constructed channel 3-years post construction; and **C** typical constructed channel 20-years post construction

locations within each site. We combined organisms and debris into a single composite sample for each site that was immediately preserved with 70% ethanol. In the laboratory, we identified organisms to genus, except for Mollusca (family), Chironomidae, Hydracarina, Oligochaeta, and Nematoda and assigned to functional feeding groups. We then used abundance data to calculate West Virginia Stream Condition Index (WVSCI) (Gerritsen et al., 2000), which is a family-level index of stream biological condition. WVSCI can be used to score sites on a scale of 0–100, with scores <68 being categorized as impaired.

OM retention was measured using artificial sticks consisting of painted dowel rods (Speaker et al., 1984). Fifty dowel rods were placed in a riffle at the upstream end of the reach. They remained on-site and the cumulative distance they traveled was measured at four intervals over 195 days (on the day of release and then after approximately 1, 2, and 6 months). The equation $T_d = T_o\, e^{-kd}$, where T_d is the percentage of released sticks remaining in transport at distance d, T_o is the original number of sticks released, and k is the instantaneous rate of retention, was used to calculate retention rate. Leaf litter packs (plastic mesh bags with 10 mm mesh size) were filled with 10 g of *Quercus palustris* (pin oak) leaves and used to quantify decomposition rates at each study site. Bags were grouped in sets of six and anchored in riffles throughout the reach length, and a total of six bag sets (for a total of 36 bags) were placed at each site. An additional set of litter bags was taken to the site and returned to the lab to calculate handling loss. One litter bag set was randomly sampled after 45, 75, 90, 120, 195, and 325 days. Bags were returned to the lab on ice, rinsed in a 250 μm sieve, and dried for 72 h to a constant mass. A subsample of 250 μg was placed into pans and incinerated to determine the ash-free dry mass (Benfield, 1996). Decomposition rates ($-k$) were determined from the slope of a linear regression relating log-transformed percent ash-free dry mass (% AFDM) to the number of days of exposure. Total OM processing was defined as the ability of a site to retain and process OM locally and was calculated by multiplying the instantaneous rate of decomposition by the instantaneous rate of retention.

Statistical analysis

We used Shapiro–Wilk's to test all variables for normality, and ln-transformed macroinvertebrate richness and biomass data for subsequent analyses. Simple correlation analysis was run on all parameters to quantify

Table 1 Summary characteristics of study sites

Site names	Site code	Mine age (years)	Mean discharge (m^3/s)	Drainage area (ha)	Latitude (DD)	Longitude (DD)
White Oak	C_WO	3	0.0159	45	38.04778	−81.52139
Argus	C_AR	5	0.0013	6	37.98972	−82.25222
Stanley Branch	C_ST	10	0.0116	33	38.08306	−81.93472
Sugartree	C_SU	10	0.0130	37	38.09007	−81.95751
Big Horse	C_BH	20	0.0030	12	38.08500	−81.89750
UT Hell Creek	R_HC	–	0.0005	7	37.73044	−82.23232
UT Lukey Fork	R_LF	–	0.0028	10	38.05944	−81.95306
UT Mud Creek East	R_ME	–	0.0011	5	38.04647	−81.91148
UT Mud Creek West	R_MW	–	0.0029	11	38.06105	−81.94331
UT White Oak	R_WO	–	0.0044	16	38.05250	−81.52278

Mine age is the approximate number of years since the mine was reclaimed. Mean discharge was calculated from four seasonal measures. Drainage area was calculated on the basis of a regional drainage area versus discharge rating curve

C constructed channel, *R* reference channel, *UT* unnamed tributary

relationships between ecosystem parameters and both conductivity and time since reclamation (constructed channels only). *T*-tests were used to test for statistical differences in ecosystem parameters between constructed and reference channels. Repeated measures ANOVAs were used to test for seasonal variation in ecosystem parameters between constructed and reference channels and to test for season x site type interactions. ANCOVAs were used to test for interactive effects of site type (i.e., reference vs. constructed) and conductivity on ecosystem parameters. Most statistical analyses were conducted using the program R Project for Statistical computing version 2.8.1. All statistical tests were evaluated at an alpha level of 0.05. To aid interpretation of our results, ecological units (EUs) (Petty & Thorne, 2005) were calculated for parameters selected as important metrics for both constructed and reference channels (i.e., functional metrics that have important meaning, such as OM decomposition rate, regardless of whether they were sampled in a lotic or a lentic environment). EUs were calculated by dividing the constructed channel mean by the reference channel mean for each parameter.

Results

Minimum daily temperature was significantly lower in reference channels, and constructed channels exhibited significantly higher discharge than reference channels (Table 2). Habitat quality was consistently higher in reference channels than in constructed channels regardless of the assessment protocol (Table 2). For example, EPA rapid bioassessment protocol (RBP) scores in reference channels were twice those observed in constructed channels (150 vs. 78) (Table 2). Nearly all vegetative characteristics also differed significantly between constructed and reference channels (Table 2).

Dissolved metals, pH, and nutrients did not differ significantly between constructed and reference channels (Table 3). Despite a lack of statistical differences in these attributes, we did observe several notable trends. First, low pH was characteristic of one of the reference channels (R_WO, unnamed tributary of White Oak Creek), and all reference channels possessed relatively low pH during the summer sampling period (Fig. 3). Second, dissolved metal concentrations tended to vary widely among the constructed channels and the highest metal concentrations were nearly always observed in constructed channels rather than reference channels (Fig. 3). For example, elevated Al, Ni, Mn, Se, and Zn were observed on the White Oak mine site relative to other mines and reference channels. Furthermore, Cd, Co, Cr, Cu, and Se tended to be elevated on the Stanly Branch mine site. Third, extremely high concentrations of nitrate were observed in the White Oak mine site (C_WO) in summer and autumn (Fig. 3). Several water quality characteristics did exhibit significant and seasonally consistent differences between constructed and

Table 2 Mean (±SE) physical habitat and vegetation attributes quantified at constructed and reference channel sites

Response variables	Constructed	Reference	*T*-Stat
Water temperature			
Max daily temp (°C)	30.7 (3.2)	31.9 (2.4)	−0.29
Min daily temp (°C)	0.15 (0.08)	0.00 (0.00)	2.28*
Mean daily temp (°C)	10.5 (0.2)	9.6 (0.7)	1.12
CV for mean daily temp	63.6 (3.5)	58.9 (4.8)	0.80
Dissolved oxygen (mg/l)	8.7 (0.5)	9.9 (1.0)	−1.11
Stream discharge			
Annual mean discharge (m^3/s)	0.009 (0.003)	0.002 (0.001)	2.16*
Habitat quality			
EPA RBP	78 (6)	150 (10)	6.22***
VA USM	4 (0)	6 (0)	−5.34***
WV FCU	3 (0)	9 (0)	−14.67***
BEHI	23 (4)	39 (3)	−3.29*
ORAM	35 (6)	61 (4)	−3.59*
Vegetation			
% Bare ground	3 (2)	29 (8)	−3.22*
% Cattail	21 (7)	0 (0)	3.06*
% Fern	0 (0)	11 (2)	−5.19***
% Forb	22 (8)	22 (5)	0.03
% Grass	22 (4)	2 (1)	5.52***
% Open water	13 (6)	0 (0)	2.04*
% Tree	1 (1)	16 (3)	−4.18**
% Vine	2 (1)	8 (2)	−3.20*
Species per km^2	0.1 (0.1)	2.5 (0.3)	−9.01***
Trees per km^2	0.3 (0.3)	9.2 (2.2)	−6.33***
% Canopy cover	4 (4)	91 (1)	−26.10***

Statistics for *t*-tests comparing attributes between site types (d.f. = 8) are also presented. See text for definitions of habitat quality

CV coefficient of variation

* $P < 0.05$, ** $P < 0.005$, *** $P < 0.001$

reference channels, including alkalinity, acidity, conductivity, total dissolved solids (TDS), calcium, magnesium, and sulfate (Table 3). All variables, except acidity, were significantly elevated in the constructed channels relative to reference streams.

TDS in the constructed channels was dominated by sulfate along with very consistent percent contributions of calcium and magnesium (Fig. 4). One of the reference sites (R_WO) exhibited water chemistry attributes consistent with acid mine drainage, which was characterized by high sulfate concentrations and an absence of bicarbonate (Fig. 4). Another reference site (R_HC) exhibited evidence of residual waste from gas drilling, which was characterized elevated concentrations of sodium and chloride, during the winter sampling season (Figs. 4). The constructed channel site, C_BH, also showed evidence of gas drilling waste (Fig. 4).

Constructed and reference channels had similar macroinvertebrate family richness (Table 4). However, constructed channels had significantly different overall macroinvertebrate community structures in comparison to reference stream channels (Table 4). Constructed channels possessed a significantly lower percentage and richness of EPT taxa and a lower percentage of shredder taxa. As a result, WVSCI scores were significantly lower in constructed channels than reference channels (Table 4). ANCOVAs revealed that differences in macroinvertebrate assemblages between constructed and reference channels were the result of significant interactive effects of conductivity and site type (Fig. 5).

Constructed channels contained highly variable larval amphibian density, whereas reference channels contained highly variable adult density (Table 4). As a result, we observed no significant differences in

Table 3 Mean (±SE) water quality attributes quantified at constructed and reference channel sites

Response variables	Constructed	Reference	*T*-stat
pH	7.4 (0.1)	6.7 (0.4)	1.56
Alkalinity (mg/l)	138 (13)	5 (1)	12.82***
Acidity (mg/l)	0 (0)	13 (5)	−4.88**
Conductivity (μS/cm)	2,197 (414)	461 (326)	3.29*
Total dissolved solids (mg/l)	25.2 (3.0)	2.7 (0.4)	6.81***
Ca (mg/l)	163 (31)	21 (15)	4.96**
Cl (mg/l)	13.8 (9.7)	11.4 (10.2)	0.45
Mg (mg/l)	154 (34)	9 (4)	6.12***
Na (mg/l)	13.2 (6.3)	23.7 (22.2)	0.09
SO_4 (mg/l)	1,008 (196)	32 (20)	7.71***
Al (mg/l)	0.1 (0.0)	0.5 (0.5)	−0.87
Ba (mg/l)	0.016 (0.002)	0.093 (0.060)	−1.72
Cd (mg/l)	0.007 (0.000)	0.008 (0.000)	−0.63
Co (mg/l)	0.011 (0.002)	0.009 (0.001)	1.00
Cr (mg/l)	0.006 (0.001)	0.009 (0.001)	−2.51*
Cu (mg/l)	0.008 (0.001)	0.008 (0.001)	0.20
Fe (mg/l)	0.10 (0.04)	0.06 (0.001)	1.03
Mn (mg/l)	0.3 (0.192)	0.4 (0.302)	0.20
Ni (mg/l)	0.034 (0.019)	0.024 (0.011)	0.48
Se (mg/l)	0.044 (0.002)	0.034 (0.005)	1.02
Zn (mg/l)	0.020 (0.008)	0.042 (0.031)	−0.45
NO_2 (mg/l)	0.04 (0.03)	3.10 (3.08)	−0.94
NO_3 (mg/l)	11.0 (10.4)	0.8 (0.2)	0.88
NH_3 (mg/l)	0.007 (0.003)	0.013 (0.002)	−1.97
TP (mg/l)	0.08 (0.04)	0.05 (0.01)	0.66

Statistics for *t*-tests comparing attributes between site types (d.f. = 8) are also presented

* $P < 0.05$, ** $P < 0.005$, *** $P < 0.001$

amphibian density or species richness between site types (Table 4). Amphibian density was uniformly low at four of the five constructed channel sites. However, a single constructed channel site (C_AR) supported amphibian densities that were an order of magnitude higher than the other constructed channels and substantially higher than three of the reference channels. Results from ANCOVAs indicate that much of the variation in amphibian assemblages could be attributed to a significant interactive effect of site type and conductivity, including total species richness ($F = 5.53$, d.f. = 7, $P < 0.001$), larval species richness ($F = 4.78$, $P < 0.01$), total amphibian density ($F = 3.06$, d.f. = 7, $P < 0.05$), and larval density ($F = 3.51$, $P < 0.01$). Site type alone was responsible for differences in the percent of lotic amphibians present at a site ($F = 2.47$, $P < 0.05$). Constructed channels supported primarily terrestrial and aquatic frogs that use lentic systems. Reference sites supported primarily aquatic salamanders that use lotic systems (Table 5).

Artificial stick retention was significantly higher in constructed channels than in reference channels for measures of mean cumulative distance traveled, percent of dowels retained within a 100 m sample site, and overall retention rate (Table 4). Leaf decomposition rates were significantly lower in constructed channels than in reference channels (Table 4). ANCOVAs indicated a significant interactive effect of site type and conductivity on leaf decomposition rates ($F = 3.42$, $P < 0.05$) (Fig. 5), suggesting that both reduced mechanical abrasion and lowered biological processing may have affected OM processing in constructed channels. Despite lower decomposition rates in constructed channels, increased OM retention rates resulted in significantly higher overall OM processing within constructed channels (Table 4). Furthermore, constructed channels averaged significantly higher dissolved organic (DOC) and total carbon concentrations than reference channels (Table 4). DOC and total dissolved carbon concentrations were highly variable

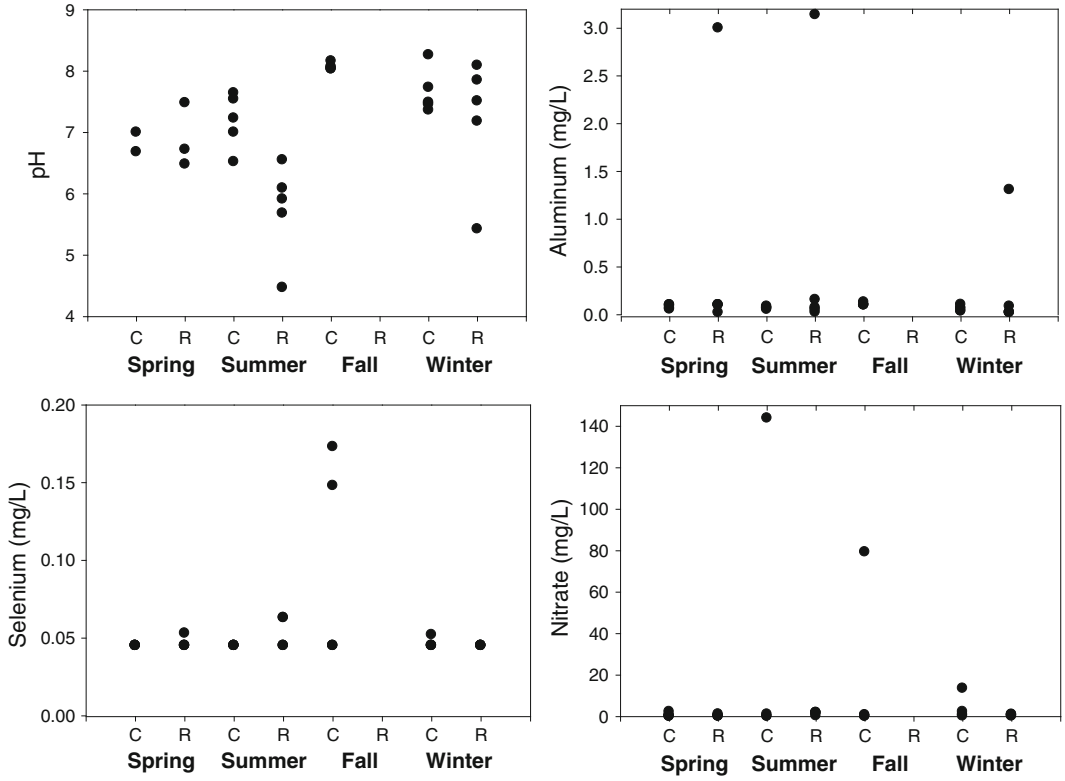

Fig. 3 Seasonal variation in pH, dissolved aluminum, dissolved selenium, and nitrate in constructed (*C*) and reference (*R*) channels

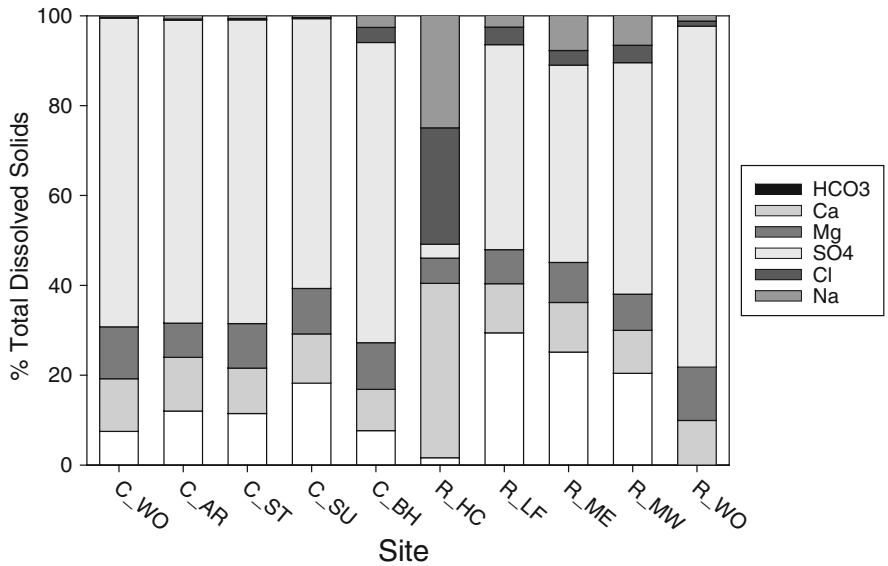

Fig. 4 Percent of mean total dissolved solids (by mass) that was composed of bicarbonate, calcium, magnesium, sulfate, chloride, and sodium for constructed (*C*) and reference (*R*) channels. Constructed channels are presented in order of age since reclamation from youngest (*C_WO*, 3 years) to oldest (*R_WO*, 20 years)

in constructed channels, whereas DOC was uniformly low in the reference channels.

Overall, we observed few significant correlations between mine age (i.e., time in years since reclamation) and various aquatic ecosystem attributes. Significant ($P < 0.05$) positive correlations were observed for percent cattail, tree density, macroinvertebrate family richness (Fig. 6), and adult amphibian

Table 4 Mean (±SE) macroinvertebrate and amphibian assemblage and OM processing attributes quantified at constructed and reference channel sites

Response variable	Constructed	Reference	*T*-stat
Benthic macroinvertebrates			
WVSCI	48 (5)	68 (8)	−2.43*
% Chironomid	58 (16)	32 (11)	1.41
% Tolerant	70 (15)	42 (13)	1.56
% EPT	5 (4)	48 (16)	−2.49*
EPT family richness	1 (0)	4 (1)	−2.67*
Total family richness	8 (2)	7 (1)	0.53
Total abundance (#/m^2)	763 (273)	213 (87)	1.81
% Collector-gatherer	74 (12	57 (9)	1.33
% Filterer	2 (2)	0 (0)	1.32
% Scraper	6 (4)	3 (3)	0.53
% Shredder	2 (2)	27 (11)	−2.55*
% Predator	8 (3)	5 (3)	0.82
% Omnivore	8 (5)	3 (3)	0.78
Amphibians			
Total species richness	3 (1)	3 (1)	−0.60
Larval species richness	2 (1)	2 (0)	0.20
Adult species richness	1 (1)	2 (1)	−1.22
Larval density (#/m^2)	19 (10)	8 (3)	0.72
Adult density (#/m^2)	5 (3)	28 (12)	−1.92
Larval biomass (g/100 m^2)	1.3 (1.0)	0.05 (0.02)	1.25
% Grassland	58 (18)	0 (0)	2.76*
% Forest	42 (5)	100 (0)	−3.00*
% Lotic	11 (3)	86 (13)	−5.11***
% Lentic	89 (3)	14 (8)	3.06*
OM			
OM decomposition rate	−0021 (0.0002)	−0.0035 (0.0009)	2.41*
% OM lost (325 days)	47 (1)	62 (9)	−1.74
OM transport distance (m/day)	0.04 (.04)	0.14 (.02)	−2.94*
% OM retained (325 days)	87 (13)	59 (9)	2.55*
OM retention rate	−0.064 (0.014)	−0.020 (0.005)	2.21*
Overall OM processing	0.013 (0.003)	0.007 (0.002)	2.56*
Total dissolved carbon (mg/l)	18.5 (2.1)	2.6 (0.3)	4.89**
Dissolved organic carbon (mg/l)	3.5 (1.0)	1.5 (0.3)	2.24*

Statistics for *t*-tests comparing attributes between site types (d.f. = 8) are also presented

* $P < 0.05$, ** $P < 0.005$, *** $P < 0.001$

species richness. Conductivity tended to decline with mine age, but this trend was not significant (Fig. 6).

Discussion

The predominant negative characteristic of constructed stream channels on reclaimed surface mines was elevated TDS and electrical conductance. Studies in this region have consistently observed strong negative effects of mining on stream conductivity (Hartman et al., 2005; Merovich et al., 2007; Pond et al., 2008; Fritz et al., 2010; Petty et al., 2010; Lindberg et al., 2011; Merriam et al., 2011). In general, where alkaline, rather than acidic, mine drainage is produced, elevated conductivity is the result of increased sulfate and bicarbonate concentrations. In other settings, mining-related increases in conductivity may be correlated with elevated dissolved metal concentrations (Petty et al., 2010). Fritz et al. (2010)

Table 5 Amphibian species expected (Exp) to occur in grassland and forest habitats in southwestern West Virginia (Green & Pauley, 1987) compared to those actually observed (Obs) as (a) adults during visual encounter surveys, as (l) during larval surveys, or as (b) both larval and adults. The preceding "c" indicates occurrence within constructed channels, and "*r*" indicates occurrence within reference channels

		Grassland		Forest	
		Exp	Obs	Exp	Obs
Aquatic salamanders					
Seal salamander	*Desmognathus monticola*			x	r.a
Eastern hellbender	*Cryptobranchus alleganiensis*			x	
Midland mud salamander	*Pseudotriton diastictus*			x	
Common mudpuppy	*Necturus maculosus*	x		x	
Northern dusky salamander	*Desmognathus fuscus*			x	r.b
Northern red salamander	*Pseudotriton r. ruber*	x		x	
Northern two-lined salamander	*Eurycea bislineata*			x	r.a
Red-spotted newt	*Notophthalmus v. viridescens*	x	c.b	x	
Southern two-lined salamander	*Eurycea cirrigera*			x	r.l
Spring salamander	*Gyrinophilus porphyriticus*			x	r.a
Terrestrial salamanders					
Cumberland Plateau salamander	*Plethodon kentucki*			x	
Four-toed salamander	*Hemidactylium scutatum*			x	
Green salamander	*Aneides aeneus*			x	
Jefferson salamander	*Ambystoma jeffersonianum*			x	
Longtail salamander	*Eurycea longicauda*	x		x	
Marbled salamander	*Ambystoma opacum*			x	
Southern ravine salamander	*Plethodon richmondi*			x	
Northern redback salamander	*Plethodon cinereus*			x	
Northern slimy salamander	*Plethodon glutinosus*			x	
Spotted salamander	*Ambystoma maculatum*			x	
Wehrle's salamander	*Plethodon wehrlei*			x	
Ambystoma species	*Ambystoma* sp.		c.l	x	
Aquatic frogs					
Bullfrog	*Lithobates catesbeianus*	x	c.a	x	
Green frog	*Lithobates clamitans melanotus*	x	c.b	x	
Pickerel frog	*Lithobates palustris*	x	c.a	x	
Northern leopard frog	*Lithobates pipiens*	x		x	
Terrestrial frogs					
Eastern American toad	*Anaxyrus a. americanus*	x	c.l		
Eastern spadefoot	*Scaphiopus holbrookii*			x	
Fowler's toad	*Anaxyrus fowleri*				
Cope's gray treefrog	*Hyla chrysoscelis*		c.l	x	
Mountain chorus frog	*Pseudacris brachyphona*			x	
Northern spring peeper	*Pseudacris c. crucifer*		c.l	x	r.a
Wood frog	*Lithobates sylvaticus*			x	

found that elevated conductivity tended to co-vary with iron and manganese concentrations in streams below valley fills associated with reclaimed surface mines in Kentucky. Fortunately, surface mine reclamation in the region of our current study appears to be successful at maintaining relatively low metal concentrations.

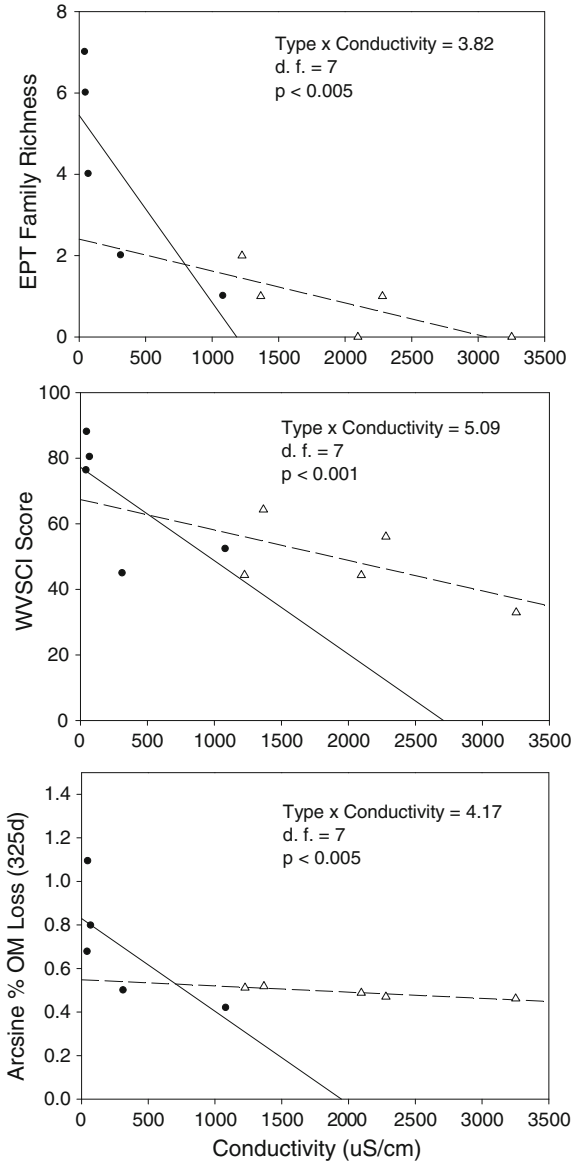

Fig. 5 Effects of site type (*open triangles* constructed channels, *closed circles* reference channels) and conductivity on Ephemeroptera, Plecoptera, and Trichoptera (EPT) family richness, WVSCI, and % OM loss from leaf decomposition packs over a 325-day period

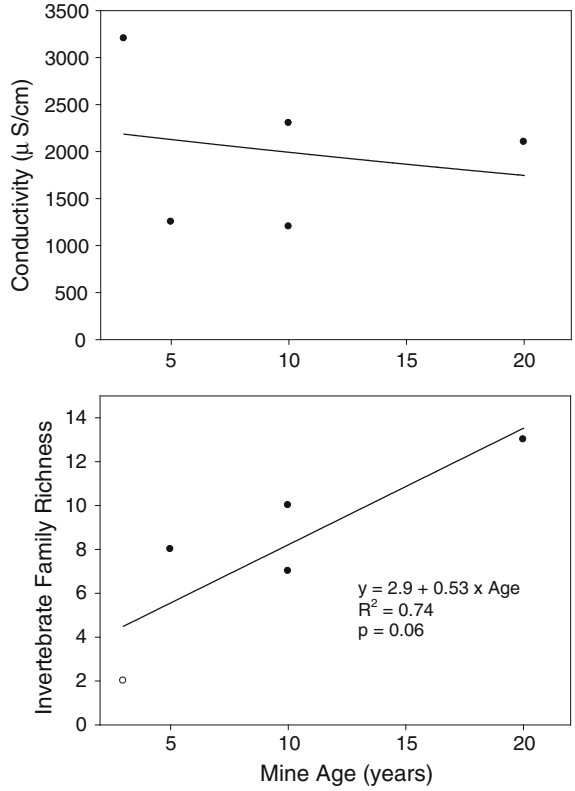

Fig. 6 Changes in conductivity (µS/cm) and invertebrate family richness in constructed channels as a function of mine age

Elevated conductivity appears to be the primary determinant of reduced biological conditions and ecosystem processes in constructed channels relative to reference channels, especially as it relates to benthic invertebrate assemblages, OM decomposition, and amphibian assemblages. Reference channels contained a significantly higher percentage of EPT than constructed channels. Additionally, a significant interaction between site type (i.e., constructed vs. reference) and conductivity was shown for WVSCI, percent EPT, EPT richness, and total invertebrate richness, suggesting an important role of elevated conductivity in determining macroinvertebrate assemblages independent of changes in physical habitat and channel substrate. The direct effect of elevated conductivity on macroinvertebrates is likely the result of major ion toxicity (Mount et al., 1997) and dissolved ion interference with the osmoregulation of macroinvertebrates (McCulloch et al., 1993).

OM decomposition rates also were significantly reduced in constructed channels relative to reference streams, a finding that is consistent with that of Fritz et al. (2010). The shift in benthic macroinvertebrate assemblages from a community supporting a large percentage of shredders to a community supporting a large percentage of collector gatherers (primarily chironomids) in response to elevated conductivity may partially explain the low rates of OM decomposition in constructed channels. Shredders play an

important role within the aquatic continuum by feeding on coarse particulate organic matter (CPOM) and converting it to fine particulate organic matter (FPOM) (Wallace et al., 1997). If the shredder community is lost or reduced, OM decomposition rates may decline as well (Fritz et al., 2010). Lower mechanical abrasion and reduced microbial activity, however, may also explain lower decomposition rates in constructed channels.

Despite reduced OM decomposition rates, significantly higher OM retention in constructed channels resulted in greater overall OM processing on-site and higher DOC. High retention rates in the constructed channels were likely the result of low gradient and high densities of emergent vegetation relative to the reference channels. Increased retention in the constructed channels means that a greater percentage of the OM entering the channel is processed locally, even if decomposition rates are lower. In fact, increased retention effectively offset reduced decomposition such that overall in situ processing of OM was equal between reference and constructed channels. Increased local processing of OM in constructed channels likely explains increases in DOC that we observed relative to reference channels. Because of the relationship between DOC and soil type and chemistry (McDowell & Likens, 1988), it is unlikely that the DOC in constructed channels originates from the soil of the reclaimed site. Constructed channel DOC is also unlikely to originate from leaf litter inputs, due to the lack of riparian trees along constructed channels. Consequently, high DOC concentrations within constructed channels most likely are originating from on-site inputs of detritus from aquatic macrophytes.

Amphibian assemblages were strongly affected by mining and reclamation, and our results are consistent with those of Wilson & Dorcas (2003) who observed significant declines in stream-dwelling salamander populations (*Desmognathus fuscus* and *Eurycea cirrigera*) in response to elevated conductivity in North Carolina streams. Both constructed and reference channels supported an average of two larval species and approximately two adult species. However, there was a significant shift in community types that can be linked to conversion from forested lotic habitat to grassland lentic habitat. Lotic and forest species, such as *Desmognathus monticola* (seal salamander) and *Desmognathus fuscus* (northern dusky salamander), were replaced by grassland-inhabiting, lentic species, such as *Hyla chrysoscelis* (Cope's gray tree frog) and *Pseudacris c. crucifer* (northern spring peeper). Constructed channels most likely benefited from close proximity to intact forest (Hecnar & M'Closkey, 1996). Constructed wetlands have been reported to be colonized by ubiquitous anurans such as gray tree frog, American toad, spring peeper, and *Rana catesbeiana* (American bullfrog) within 2 years of creation (Pechmann et al., 2001). The colonization by amphibians shortly after wetland establishment is consistent with patterns found in accidentally formed and constructed wetlands (Pollio, 2005).

During the process of MTR-VF mining and reclamation steep, forested headwater catchments are converted to unforested catchments with rolling terrain and wetland-like aquatic features. Calculation of "EUs" (Table 6), which represent a ratio between the functional value of constructed and reference channels, provides an objective means quantifying the extent to which reclaimed channels compare to the disturbed reference channels. First, it is clear that several important functional characteristics of headwater streams may be permanently lost as a result of the surface mining/reclamation process. These functions include the ability to support sensitive stream-dwelling salamanders and mayflies and the ability to support high rates of OM decomposition (Table 6). Second, it is clear that several important characteristics are either unaffected or accentuated following mine reclamation. Improved functions include the ability to support a productive assemblage of lentic amphibians, the ability to retain and process OM locally, and the ability to produce and retain DOC (Table 6).

Our results suggest the following recommendations with regard to managing mining-related impacts to streams. First, our results highlight the need for improved mine spoil handling during reclamation so as to minimize TDS generation, including approaches that reduce the contact time between TDS-generating spoils and water. Furthermore, construction of sulfate-reducing wetlands as part of mine reclamation may be a way to maximize aquatic habitat functions on reclaimed mines and reduce TDS concentrations. Second, previous studies have emphasized the connectivity of upstream ecological processes to downstream ecosystem function and value (Gomi et al., 2002; Wipfli et al., 2007). For the reclaimed headwaters to be of maximal value at the watershed scale, care

Table 6 Constructed channel means (±SE), reference channel means, ideal reference means (based on reference channels without elevated conductivity), EU ratios, and EU_I ratios (calculated using ideal reference means) for important biological parameters

Response variables	Constructed	Reference	Ideal reference	EU ratio	EU_I ratio
Larval amphibian biomass (g/100 m^2)	1.33 (1.03)	0.05 (0.02)	0.07 (0.02)	**26.60**	**19.00**
OM retention rate	−0.0644 (0.0139)	−0.0199 (0.0049)	−0.0189 (0.0021)	**3.24**	**3.41**
Mean DOC (mg/l)	3.51 (0.94)	1.51 (0.28)	1.73 (0.14)	**2.32**	**2.03**
Overall OM processing × 100	0.013 (0.003)	0.007 (0.002)	0.009 (0.003)	**1.86**	**1.44**
% Lentic amphibians	89 (8)	54 (8)	61 (11)	**1.65**	**1.46**
Total invertebrate richness	8 (2)	7 (1)	8 (1)	1.14	1.00
Total invertebrate biomass (g/m^2)	32 (19)	35 (15)	44 (23)	0.92	0.72
WVSCI score	48 (5)	68 (8)	81 (3)	**0.71**	**0.59**
Decomposition rate	0.0021 (0.0002)	0.0035 (0.0009)	0.0045 (0.0011)	**0.57**	**0.44**
EPA RBP	78 (6)	150 (10)	134 (1)	**0.52**	**0.58**
EPT richness	1 (0)	4 (1)	6 (1)	**0.25**	**0.17**
% EPT	5 (4)	48 (16)	72 (11)	**0.10**	**0.07**
% Lotic amphibians	5 (3)	86 (13)	99 (1)	**0.06**	**0.05**

EU ratios greater than 1.0 represent conditions where the values observed in the constructed channels exceeded those observed in the reference channels. Response variables are sorted such that variables with the high positive response to mining are listed first. Bold values reflect EU ratios for biological response variables that differed significantly ($P \leq 0.05$) between constructed and reference channels

needs to be taken to ensure that the constructed systems are connected by surface and/or subsurface flow to aquatic ecosystems downstream. Third, the shift in local amphibian and macroinvertebrate assemblages from lotic, sensitive taxa to lentic, generalist taxa may become problematic as the cumulative effects from mine to mine are considered at a regional scale (Merriam et al., 2011). Although prevention of species loss at a local scale may not be possible, consideration must be given to protect portions of mined watersheds to act as source populations for species re-colonization (Lowe et al., 2006; Pond et al., 2008). Finally, some level of elevated TDS from new mines must be expected and considered within the context of existing and future sources (Merriam et al., 2011). By preserving a percentage of tributaries as sources of dilute water, cumulative downstream changes in water chemistry may be avoided (Saunders et al., 2002). This may be an especially pertinent solution, because mining may not be the only stressor to local watersheds within the region (Merriam et al., 2011).

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