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Impact of coal mining on stream biodiversity in the US and its regulatory implications

Xingli Giam^{1*}, Julian D. Olden² and Daniel Simberloff¹

Coal mining is a major cause of land-use change in the US, and according to the Energy Information Administration it is expected to remain a key part of the national electricity portfolio until at least 2040. It is therefore crucial to understand the environmental impact of coal mining. Although a scientific consensus has emerged that coal mining negatively affects water quality, a quantitative synthesis of biodiversity impacts is currently lacking. Here, we show that mining under current federal statutes—the 1972 Clean Water Act and the 1977 Surface Mining Control and Reclamation Act—has negative implications for freshwater biota. Streams affected by coal mining averaged one-third (32%) lower taxonomic richness and one-half (53%) lower total abundance than unmined streams, with these impacts occurring across all taxa investigated thus far (invertebrates, fish, and salamanders). Even after post-mining reclamation, biodiversity impacts persisted. Our investigation demonstrates that current US regulations are insufficient to fully protect stream biodiversity.

The Energy Information Administration (EIA) projects an overall increase in US coal production from 740 million short tons (MMst) in 2016 to 861 MMst by 2040 without the Clean Power Plan, assuming intermediate technological, demographic and economic trends and persistence of other prevailing laws and regulations¹. With the Clean Power Plan, coal production is projected to increase first from 740 MMst to 835 MMst by 2020, then decrease to 619 MMst by 2040. These scenarios (and others that assume various economic and technological trajectories) indicate coal mining will likely remain an important driver of US environmental change.

Coal mining in the US is currently governed primarily by two federal statutes: the 1972 Clean Water Act (CWA) and 1977 Surface Mining Control and Reclamation Act (SMCRA)^{2,3}. Section 404 of the CWA governs discharge of dredged or fill material, whereas section 402 of the CWA regulates discharge of all other pollutants. SMCRA and its regulations complement the CWA by establishing operational standards to minimize surface impacts of both surface and underground mines. For example, before starting operations, mining companies must submit a permit application explaining plans to “prevent material damage to the hydrologic balance outside the permit area,” “minimize disturbances and adverse impacts ... on fish, wildlife, and related environmental values” and so on.

Regulations under SMCRA and CWA, however, appear largely ineffective to mitigate coal mining impacts on water quality. Negative consequences of mountaintop mining–valley fill (MTM–VF), a type of surface mining that has greatly altered central Appalachian landscapes^{3–5}, are particularly well-studied. In MTM–VF, vegetation and soil on mountain ridges are cleared and rocks overlying coal seams are removed by explosives to access underlying coal. Some waste rocks are stored for reclamation following completion of mining, but excess rocks are dumped in valleys (forming valley fills). In a sample of 15 MTM–VF streams in West Virginia, concentrations of Se in 13 streams exceeded EPA allowable limits⁶ and would lead to toxic bioaccumulation in fish and wildlife⁷. Concentration of ions including Ca^{2+} , Mg^{2+} , SO_4^{2-} and HCO_3^- are also elevated downstream of valley fills^{3–5,8–10}. The extent of these impacts far exceeds the local footprint of individual mines; solutes are transported and accumulate downstream, even after reclamation is completed¹¹.

Although relatively fewer studies were published on water-quality impacts of other mining methods and in other regions under SMCRA and CWA, evidence suggests that water-quality impacts likely extend beyond MTM–VF and central Appalachia. For example, stream conductivity increased downstream of treated discharges from an underground mine in Pennsylvania⁹. In Illinois, a surface mine was found to discharge effluent with pollutant levels that exceeded acidic monthly average/daily maximum limits, violating its National Pollutant Discharge Elimination System (NPDES; section 402 of the CWA) permit¹².

Given deteriorating water quality, we can expect coal mining to affect stream biodiversity. Indeed, many studies have individually investigated biotic impacts in various watersheds and streams across the US^{8,9,13–17} (Fig. 1). However, recorded effects vary: some studies revealed lower taxa richness or density in mine-affected streams^{9,13–15}, whereas others did not^{8,16,17}. Because each study examined impacts under a narrow operational context (active versus reclaimed mines; underground versus MTM–VF) in one or a few watersheds, and only for one or two taxonomic groups, a quantitative meta-analysis is needed to synthesize findings of individual studies to facilitate comparisons across mining contexts and taxa for a more-comprehensive understanding of biodiversity impacts.

Here, we conduct a quantitative meta-analysis to elucidate the impact of coal mining on richness and abundance of different taxonomic groups across different mining systems under existing regulations according to SMCRA and CWA. We also identify watersheds of conservation concern where both conservation risk owing to current and near-future coal production and conservation need in terms of endemic and threatened fish species richness are highest. Last, we discuss how existing regulations should be strengthened to minimize biodiversity impacts of coal mining and suggest various strategies to garner broad public support for this endeavour.

Results

Types of coal mining, biodiversity and reclamation efficacy. Our meta-analysis examined taxonomic richness and total abundance in streams affected by mining in the SMCRA and CWA era (post-1977) versus unmined streams in the US. We conducted a systematic

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Fig. 1 | Coal mining and impacted stream biodiversity. **a**, Mountaintop mining near Whitesburg, Letcher County, Kentucky. **b**, Runoff from a valley fill in Kentucky. **c**, Spring salamander (*Gyrinophilus porphyriticus*), a species extremely sensitive to MTM-VF. **d**, Emphemerellid mayfly nymphs like the small blue-winged olive (*Drunella cornutella*) are heavily impacted by elevated ion concentrations in streams affected by MTM-VF. **e–g**, Diamond darter (**e**, *Crystallaria cincotta*), tuxedo darter (**f**, *Etheostoma lemniscatum*) and Kentucky arrow darter (**g**, *E. spilotum*) are federally T&E species impacted by coal mining. Photographs courtesy of www.mountainroadshow.com (**a**), Matt Wasson/Appalachian Voices (**b**), Todd Pierson (**c**), Ben Henricks (**d**), Conservation Fisheries (**e**), Todd Amacker (**f**), and Matt Thomas/Kentucky Dept. of Fish and Wildlife Resources (**g**).

literature search for empirical studies that surveyed biotic assemblages in multiple mine-affected and unaffected stream sites (hereafter, mined versus unmined sites). We retained studies that presented means and standard deviations of richness and/or abundance in the original paper. If data were unavailable in the paper, we requested them from authors and included these studies if data were provided. In total, we obtained data for 13 studies from which 21 comparisons of taxonomic richness and 17 comparisons of total abundance in mined versus unmined sites were analysed (Table 1; Supplementary Tables 1–3). We used hierarchical random effects meta-analysis to summarize effect sizes for the full dataset and MTM-VF mining, and hierarchical mixed effects meta-analysis to estimate effect sizes across different taxonomic groups and reclamation statuses (see Methods).

Overall, taxonomic richness and abundance were significantly lower in mine-affected sites (Fig. 2). Richness and abundance averaged 32% (95% confidence interval: 24–40%) and 53% (10–75%) lower, respectively, in mined sites than unmined sites. Mining affected all taxonomic groups (invertebrates, fish and salamanders) negatively; invertebrate richness declined on average by 32% (19–42%), fish richness by 39% (19–54%) and salamander richness by 28% (0–48%). In the one study that examined underground mining impacts, richness and abundance of mined sites were 27% and 41% lower than unmined sites, respectively. These values were comparable to MTM-VF impacts (33% (23–41%) and 54% (7–77%) reduction in richness and abundance, respectively).

Table 1 | Number of studies (Std) and comparisons (Comp) across states, mine types, taxa and reclamation status for taxonomic richness and total abundance analyses.

	Overall		Richness		Abundance	
	Std	Comp	Std	Comp	Std	Comp
Location						
West Virginia	8	26	7	15	6	11
Kentucky	5	10	5	5	5	5
Pennsylvania	1	2	1	1	1	1
Mine type						
MTM-VF	12	36	11	20	10	16
Underground	1	2	1	1	1	1
Taxa						
Aquatic invertebrates	8	24	7	14	6	10
Fish	3	10	3	5	3	5
Salamanders	2	4	2	2	2	2
Reclamation status						
Ongoing/active mining	9	29	8	16	7	13
Reclaimed	5	9	5	5	4	4

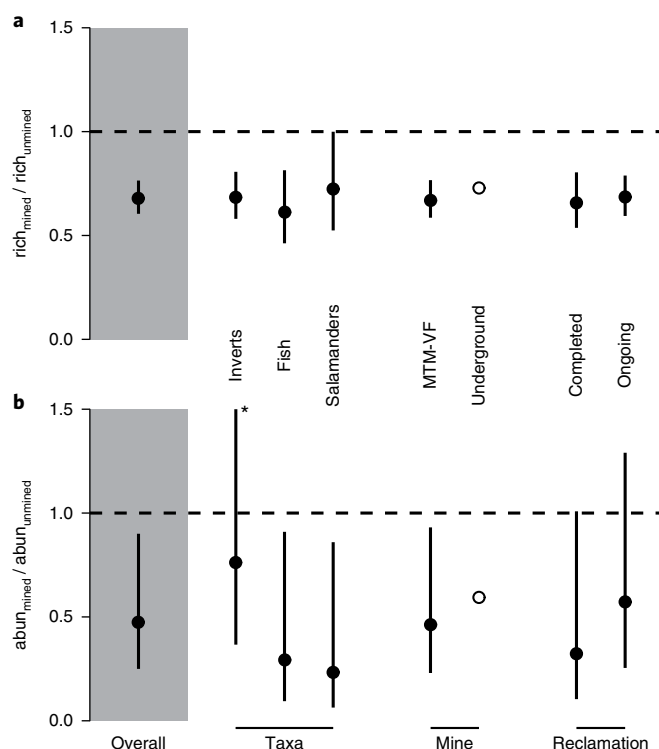


Fig. 2 | Effects of coal mining on stream biodiversity. **a**, **b**, Mining impact is quantified by the ratio between taxonomic richness (**a**) and total abundance at mined versus unmined sites (**b**). Solid points are mean effects, whereas lines denote 95% confidence intervals from hierarchical random- and mixed-effects meta-analyses. The horizontal dashed line at 1.0 is the point at which richness (**a**) and abundance (**b**) are equal in mined and unmined streams. The location of mean effects below this line indicates that, on average, richness and abundance are lower in mined streams than in unmined streams. We did not perform a meta-analysis for underground mining because only one study/comparison was available; open points denote observed effect sizes. The upper confidence bound of effect size for aquatic invertebrate abundance is 1.58 but it is truncated at 1.5 for presentation purposes, denoted by an asterisk.

Mining impacts on richness were less variable across studies than impacts on abundance based on the magnitude of among-study variance (Supplementary Table 4).

A central aim of SMCRA is to ensure mining companies complete reclamation (that is, rehabilitation of mined areas to pre-mined condition) by requiring them to post a performance bond. However, it is currently unclear whether reclamation aids biotic recovery. Our meta-analysis showed that even when reclamation is completed, mined sites supported on average only 66% (54–80%) of the richness and 32% (10–101%) of the abundance in unmined sites (Fig. 2).

Different studies inevitably employ different protocols to enumerate biodiversity. Although all studies standardized sampling methods across mined and unmined sites, only some addressed potential biases from differences in sampling intensity or completeness by subsampling to roughly equal abundances (for example, in refs^{14,18}) or by conducting occupancy modelling¹⁹. To examine whether differences in sampling completeness may bias effect sizes, we reanalysed all studies that provided site-abundance data^{15,20–23} to re-estimate richness using coverage-based rarefaction²⁴. Effect sizes based on rarefied richness only slightly exceeded those based on observed richness (Supplementary Table 5; Supplementary Fig. 1a). Using rarefied richness effect sizes (instead of those based on observed richness) did not affect our results substantively (Supplementary Fig. 1b), indicating that variation in sampling completeness is unlikely to bias our conclusions.

Results of meta-analyses may be biased when smaller studies or insignificant results are less likely to be published (that is, publication bias²⁵). We assessed the potential for publication bias by visually inspecting a funnel plot, which plots a measure of the certainty of effect sizes ($1/\sqrt{v}$; higher value = higher certainty) against effect sizes (L) (ref.²⁵). In the absence of publication bias, this scatterplot will be roughly symmetrical around the overall and group mean effect sizes. Studies with high certainty will be more clustered around the mean effect size, whereas low-certainty studies will have more dispersed effect sizes. The funnel plot for abundance was roughly symmetrical, whereas the richness plot showed a slight asymmetry, driven largely by a single study²⁰ (Supplementary Figs. 2,3). Excluding this study did not affect our effect size estimates substantially (Supplementary Fig. 4). These results indicate little potential for publication bias to affect our overall conclusions.

Watersheds of conservation concern. As coal mining is poised to continue until at least 2040¹ and is likely to expand to previously unmined lands as current mines deplete, we identified the watersheds most likely to incur deleterious effects of mining on biodiversity. Watersheds in which ongoing and likely near-future (~10–15 years) coal production is greatest and that harbour the most endemic and federally threatened/endangered (T&E) aquatic species are likely to suffer the greatest and most irreplaceable biodiversity declines and thus deserve the greatest regulatory scrutiny. We identify such watersheds as ‘watersheds of conservation concern’ or WCC. Watersheds in our analysis are defined as USGS 8-digit Hydrological Unit Code (HUC-8) units.

Using data from the EIA²⁶, we confirmed that county-level coal production in each decade (from 1983–2016) is correlated with total production and production from new mines in the following decade (Supplementary Fig. 5). Based on these findings, we use coal production from 2005–2016 (divided by watershed area to control for watershed size differences) to represent present and likely near-future intensity of coal mining in a watershed. Using coal production data that span more than one decade (1994–2016 or 1983–2016) did not substantially change our findings (because production was correlated across successive decades). Our analysis focused on freshwater fishes because they are a highly diverse

and economically important group with well-established watershed-level distributions²⁷. We calculated for each watershed two indices of biodiversity value: (i) endemism-weighted total species richness and (ii) endemism-weighted T&E species richness. Both indices consider the irreplaceability of a given species by assigning more weight to species with smaller native ranges (ref.²⁸; see Methods).

We define watersheds of the highest conservation concern (WCC1) as those ranked in the top 5% in both coal production and endemism-weighted total richness (Fig. 3; Supplementary Table 6). Watersheds included in WCC1 are the Elk in West Virginia (HUC-8 05050007), Upper Clinch (06010205) and Powell (06010206) in Tennessee and Virginia, Upper Cumberland (05130101) in Tennessee and Kentucky, Locust (03160111) and Upper Black Warrior (03160112) in Alabama, and Upper Mississippi-Cape Girardeau (07140105) in Illinois and Missouri (Fig. 3). Watersheds of high conservation concern (WCC2) are defined as those ranked in the top 10% in both coal production and endemism-weighted total richness or T&E richness but not already in WCC1. We identified a total of 30 WCC2 watersheds, and they are generally located near WCC1 watersheds. WCC2 watersheds include the North Fork, Middle Fork and South Fork Kentucky watersheds in central Appalachia; Mulberry, Sipsey Fork and Cahaba in southern Appalachia; and the Lower Yampa, Silver and White in the Colorado Plateau. WCCs identified based on total richness were largely congruent with those based on T&E richness except for WCCs in the Colorado Plateau, which are characterized by high T&E (but not total) richness, and some southern Appalachian WCCs, which have high total richness but relatively low T&E richness.

Federally T&E species of particular conservation concern include the diamond darter (*Crystallaria cincotta*), which is also critically endangered globally, the tuxedo darter (*Etheostoma lemniscatum*), the Kentucky arrow darter (*E. spilotum*) (Fig. 1e–g), and the blackside dace (*Chrosomusumberlandensis*). The diamond darter is restricted to the Elk watershed (WCC1), which is currently affected by pollution and habitat loss from coal mining and other human activities²⁹. The tuxedo darter, a globally vulnerable species known from only a single 19 km stream reach in the South Fork Cumberland watershed (WCC2), is threatened by sedimentation and water pollution from mining³⁰. The Kentucky arrow darter, which is restricted to North, Middle and South Fork Kentucky watersheds, and the globally endangered blackside dace, whose range includes the Upper Cumberland and Powell watersheds (both WCC1), have declining populations vulnerable to siltation and acid mine drainage^{31,32}. Finally, species not formally protected but that are nevertheless restricted to intensively mined watersheds (for example, emerald darter; *E. baileyi*) may be vulnerable to undocumented declines and should be regularly monitored.

Discussion

Biodiversity impacts of coal mining. Our study demonstrates that coal mining under current regulations is detrimental for stream biodiversity across multiple taxa. The variation in individual study effect sizes is likely due to differences in taxonomic resolution (genus versus family level), sampling techniques, distance from study site to upstream mining and/or stream size across different studies. As indicated by our rarefaction analyses on studies with available data, effect sizes estimated in our meta-analysis are robust to imperfect sampling of taxa within each site.

Multiple mechanisms could explain observed declines in biodiversity richness and abundance associated with coal mining. In MTM-VF, waste rock placed along valley bottoms buries miles of stream, reducing the amount of habitat available for stream biota^{3–6}. Treated effluents from underground mines⁹ and runoff through valley fills⁵ elevate aqueous solute concentrations in downstream ecosystems^{11,33}. Some of these solutes, such as Se, Mg²⁺, K⁺, HCO₃[–]

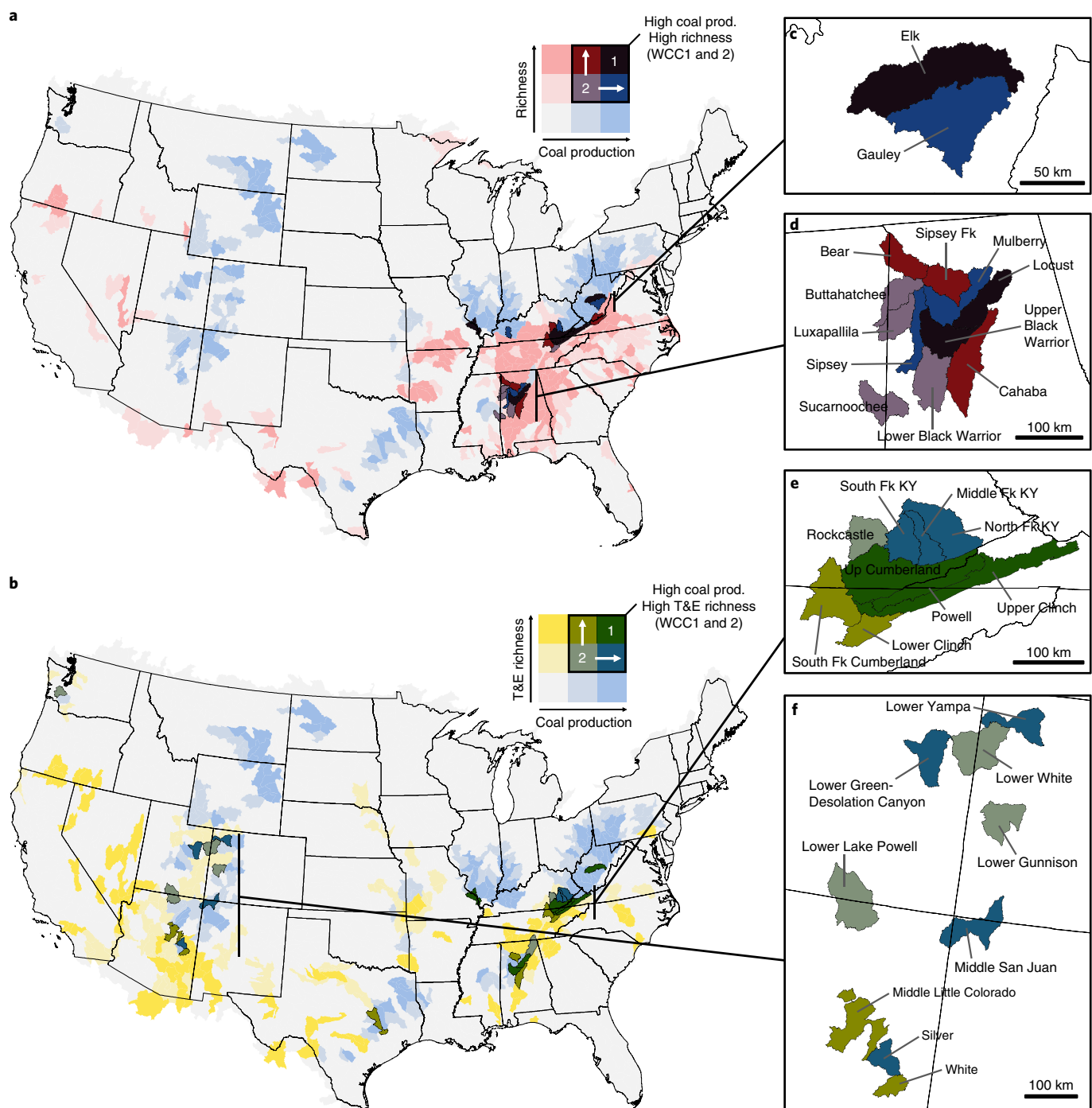


Fig. 3 | Watersheds of highest (WCC1) and high (WCC2) conservation concern. **a,b**, Based on coal production and endemism-weighted total fish species richness (**a**) or T&E fish species richness (**b**). Bivariate choropleth maps show the overlap between coal production (light to dark blue hues from left to right represent increasing coal production, that is, <90th percentile; 90–95th percentile; 95th–100th percentile) and richness (light to dark red/yellow hues from bottom to top represent corresponding increases in total richness and T&E richness) across watersheds. Colours of cells at the top-right corner therefore represent watersheds ranked in the top 5% (WCC1) and 10% (WCC2) in both coal production and fish richness. **c–f**, Appalachian watersheds are of particular importance owing to high total and T&E richness (**c–e**), whereas high T&E richness drives prioritization of Colorado Plateau watersheds (**f**).

and SO_4^{2-} , are directly toxic to stream biota^{5,7}. In addition, elevated SO_4^{2-} also leads to increased production of bisulfide (HS^- ; ref.³), which is toxic to aquatic macrofauna and macrophytes³⁴, and promotes eutrophication, reducing O_2 availability for aquatic fauna³. Increased bulk ionic concentrations in stream water may negatively affect aquatic biota by disrupting osmoregulation^{3,35}.

The net acidity of mine drainage depends on the geology of the coal and overlying strata, and mitigation strategies employed during mining^{5,36,37}. In the central Appalachian region (where original studies analysed in our meta-analysis were located), mine drainage is neutral to alkaline in many areas because H^+ ions produced from pyrite oxidation are neutralized by carbonates naturally occurring

in the overburden or added to the valley fill⁵. Indeed, all studies analysed here sampled sites that had neutral to alkaline pH. We are unaware of any study that examined the relative biodiversity impact of acid versus neutral-alkaline drainage from SMCRA-era mines. However, untreated acid mine drainage (AMD) from pre-SMCRA abandoned mines resulted in complete or near-complete extirpation of fish in many sites^{38,39} and moderate/large reductions (~48–95%; refs^{40–42}) in aquatic invertebrate richness, indicating that the impact of AMD is likely equal to or greater (for fishes) than neutral-alkaline mine drainage.

Section 404 of the CWA requires companies to undertake compensatory mitigation (that is, restore, recreate and preserve other waters) to offset unavoidable impacts from valley fills in MTM-VF. A recent comprehensive review of mitigation projects in central Appalachia revealed fundamental problems including inadequate regulatory requirements for assessing mitigation success, frequent mismatch between impacted sites (mostly intermittent or ephemeral) and mitigated sites (mostly perennial) and importantly, the general inefficacy of such projects in improving biotic integrity and water quality⁴³. This indicates that biodiversity impacts described in our analysis are rarely fully compensated by off-site mitigation activities.

Our findings must be interpreted in the light of limitations associated with our data and analysis. First, apart from one study that investigated underground mining impacts in Pennsylvania, all other studies examined MTM-VF, which is a specific type of surface mining currently restricted to the mountainous landscapes of central Appalachia. We found no studies investigating SMCRA-era mining in other regions, possibly because (i) MTM-VF became common starting in the 1970s (hence overlapping with SMCRA) and/or (ii) researchers perceived MTM-VF to be less benign than other types of mining, thus prioritizing its study. Effect sizes estimated in our analyses are therefore generalizable only within the central Appalachian region. Nevertheless, our results remain important because central Appalachia is a hotspot for both coal production and endemic and T&E species (as indicated by our WCC analysis). To improve our understanding of mining impacts more broadly, additional studies examining possible mining impacts in other regions (for example, the western region, which supplies the majority of US coal) are urgently required.

Second, all studies included in our meta-analysis were ‘control-impact’ (CI) studies that compared control (unmined) with impacted (mined) sites in a single time-period after the impact has occurred. Unlike ‘before–after control-impact’ (BACI) and ‘BACI-paired series’ (BACIPS) experimental designs, CI studies may confound natural spatial variability with the true effect of the impact investigated⁴⁴. Because it is not possible to clearly remove spatial bias from the effect size with CI ‘after-only’ data, our effect size estimates inevitably incorporate spatial variability in site conditions. Importantly, this issue of natural spatial variability demonstrates the need for mining companies (or state regulators) to collect reliable pre-mining data, a requirement of the recently repealed Stream Protection Rule.

Improving stream protections. Improvements in regulations and their enforcement can help reduce the impact of coal mining on stream biodiversity and water quality. In December 2016, the Office of Surface Mining Reclamation and Enforcement (OSMRE) issued the Stream Protection Rule (SPR⁴⁵), which revised regulations implemented under SMCRA effective 19 January 2017. The SPR had aimed to reduce coal-mining impacts on water quality and biodiversity by requiring companies to collect pre-mining, mining and post-mining/reclamation water quality and biodiversity data at regular time intervals to facilitate remediation of impacts and assessment of reclamation efficacy. In addition, under the SPR, prohibition of durable rock fills would likely have increased stability of valley fills and reduced leaching of mineral ions in MTM-VF

landscapes in Appalachia⁴⁵. However, in February 2017, Congress enacted a joint resolution to repeal the SPR, H.J.Res. 38, which President Trump then signed.

Tightening current regulations starting from the reinstatement of the SPR would likely help reduce the current impact of coal mining on stream biodiversity owing to the prohibition of durable rock fills and more stringent monitoring requirements. Going beyond the SPR, dumping of any mining waste should be prohibited in all streams to preserve habitats and to minimize downstream pollution, especially because compensatory mitigation under section 404 of the CWA has been shown to be largely ineffective⁴³. In addition, better implementation and/or enforcement of existing regulations such as NPDES effluent limits and the cumulative hydrological impact assessment (CHIA) is required^{12,46}. Further, because coal mining affects a wide range of taxa even after reclamation, more research on improving reclamation outcomes should be conducted to inform more-effective reclamation standards and/or enforcement mechanisms in both surface and underground mining operations. Lastly, we suggest MTM-VF permits not be granted once the total mined area exceeds 6.3% of the watershed; this is the threshold at which the average stream is likely to become biologically impaired as defined by multiple biotic indices with $\geq 95\%$ confidence⁴⁷.

Public support for stream protections. Garnering broad public support for stream protections will help drive policies and regulations that can better protect stream biota. Issues that may attract broad support among the public across political beliefs and affiliations include improving public health outcomes and preserving opportunities for recreational fishing and game hunting. Improving public health outcomes is attractive across the political spectrum because it reduces government spending while increasing the well-being of vulnerable communities. Coal mining is linked to increased human mortality from lung cancer, chronic health problems and kidney disease⁴⁸. The human cost of coal mining is six times greater than its economic benefits, a conservative estimate that does not include increased federal expenditure on programmes such as Medicaid⁴⁸. Tightening environmental and health protections will internalize current external costs of human health for mining companies, reducing governmental subsidies and spending for health programmes while leading to better environmental and human health outcomes.

It is also opportune for conservation organizations to engage anglers and hunters more actively and effectively, the majority of whom are Republican, conservative, and vote in every election⁴⁹. Within this community, bipartisan support is very strong for CWA protections to wetlands and waterways (support from 94% of Democrat, 78% of independent, and 73% of Republican members⁴⁹). Because legislators respond to their constituents, engaging anglers and hunters to register their concerns for environmental protections with their senators or representatives may help drive bipartisan legislative support. This idea is not new; environmental non-profits such as the Sierra Club and hunter/angler-focused groups such as Backcountry Hunters and Anglers and Trout Unlimited have been communicating with their members and the public on the importance of conserving public lands and streams. However, for more-effective engagement, a deeper understanding of people’s attitudes toward environmental protections is necessary. For example, no study has examined whether recreationists are more concerned with protecting the environment they use or all environments in general. If a study reveals that it is the former, an effective engagement strategy might include educational campaigns focusing on how pollution in headwater streams may affect downstream hunting/angling habitats.

The US has long been a global leader in environmental protection. Unfortunately, coal mining under current laws and regulations continues to impact stream biodiversity and water quality. It is now more

urgent than ever to acknowledge these impacts and, more importantly, to find broad public and political support to improve environmental safeguards that will ultimately benefit our communities.

Methods

Meta-analysis of coal-mining impacts. *Data.* We used Web of Science to search for relevant articles from all years using the query, TS = ((coal) AND (mining* OR mine* OR effluent* OR waste* OR wastewater* OR debris* OR fill*) AND (biodiversity* OR fish* OR macroinvertebrate* OR invertebrate* OR mollusc* OR gastropod* OR snail* OR mussel* OR fauna* OR plant* OR animal* OR flora* OR amphibia* OR vertebrate*)) AND ((communit* OR assemblage* OR richness* OR diversity* OR biomass* OR abundance* OR producti*) AND (stream* OR aquatic* OR creek* OR river* OR headwater* OR freshwater* OR lake*)). We read all studies and searched their citations to obtain other relevant papers. We included studies that recorded means and standard deviations of taxonomic richness and/or abundance in multiple sites affected by mining in the SMCRA and CWA era (post-1977) and multiple unmined sites. Data were requested from authors if means and standard deviations were unavailable from the published paper, and studies were included if authors provided their data. Mined sites were defined by authors as being situated in mined watersheds and/or receiving mine drainage, whereas unmined sites were in unmined watersheds or upstream of mines/mine drainage. A total of 13 studies met our criteria and were analyzed in our meta-analysis.

Studies were conducted in West Virginia (eight studies), Kentucky (five studies) and Pennsylvania (one study); numbers do not equal thirteen because one study investigated impacts in both Kentucky and West Virginia. In total, twelve studies investigated MTM-VF impacts; only one examined the impact of underground mining. Most studies focused on aquatic invertebrates (eight studies), followed by fish (three studies) and stream salamanders (two studies). Nine studies examined streams affected by mines that were active or undergoing reclamation, whereas five examined streams affected by reclaimed mines. Numbers do not equal thirteen because one study examined both types of stream (Table 1). No studies explicitly examined the impact of mines operated during the SMCRA era and then abandoned without reclamation.

To compare biodiversity values in mine-affected versus unaffected streams, we calculated the log-response ratio effect size (L ; ref.⁵⁰), which is the ln-transformed ratio of the mean richness or abundance in mine-affected streams ($\bar{x}_m$) versus unaffected streams ($\bar{x}_u$):

$$L = \ln \left(\frac{\bar{x}_m}{\bar{x}_u} \right) \quad (1)$$

Therefore, when the average richnesses or abundances in mined and unmined sites are the same, $L = \ln(1) = 0$. When mined areas contain only half the richness/abundance of unmined areas, $L = \ln(0.5) = -0.69$; whereas when mined areas contain twice as many species as unmined areas, $L = \ln(2) = 0.69$. The variance of L , v , was calculated as:

$$v = \frac{(s_m)^2}{n_m \bar{x}_m^2} + \frac{(s_u)^2}{n_u \bar{x}_u^2} \quad (2)$$

where s_m and s_u are the standard deviations of x_m and x_u , respectively; and n_m and n_u are the sample sizes of mined and unaffected sites, respectively. We confirmed that log-response ratio effect size is appropriate for our analysis (versus mean difference and standardized mean difference, Hedges's g) by plotting candidate effect sizes against richness and abundance (Supplementary Fig. 6).

Different studies used different protocols to enumerate biodiversity. Some studies subsampled individuals while one study used occupancy modelling to account for imperfect sampling across sites. To examine whether unaccounted differences in sampling completeness may bias effect sizes, we re-estimated (rarefied) richness values for studies that originally reported observed richness values (and raw abundance data) using coverage-based rarefaction⁵⁴ implemented in the iNEXT package⁵¹ in R⁵². Coverage-based rarefaction standardizes different samples (sites) within a study to equal sampling completeness. We prefer coverage-based rarefaction to size (abundance)-based rarefaction because the former is less likely to result in biased richness comparisons between samples⁵⁴. Our analysis showed that effect sizes based on rarefied richnesses were only slightly higher than those based on original richnesses (Supplementary Table 5; Supplementary Fig. 1a), and using rarefied richness effect sizes (observed richness effect sizes) did not affect our results (Supplementary Fig. 1b).

We summarized effect size (L) of taxonomic richness and of abundance for all comparisons/studies (that is, overall effect size) and for MTM-VF mines using hierarchical random-effects meta-analysis. The effect size for each individual taxonomic group (invertebrates, fish, salamanders) and reclamation status (reclamation completed versus ongoing reclamation/active mining) was estimated using hierarchical mixed-effects meta-analysis, which pools among-study and among-comparison variance across all groups. The mixed-effects

model is preferable to fitting a random-effects model for each group individually because the variance component is estimated across a larger number of studies/comparisons⁵³. This approach is reasonable because the amount of heterogeneity appears to be similar across groups based on the funnel plots⁵³ (Supplementary Figs. 2,3). We did not fit a mixed-effects model to mine type because there was only one comparison involving underground mines; a random-effects model was fitted to MTM-VF data and the estimated effect size from this analysis was compared to the observed effect size for the single underground mine comparison.

Our hierarchical random-effects and mixed-effects models incorporate random effects at the comparison-level as well as the study-level to account for potential dependence (that is, clustering) among multiple comparisons within each study, if present. The first (within-study) level of both models can be expressed as:

$$L_{ij} = \theta_{ij} + e_{ij} \quad (3)$$

where L_{ij} is the effect size of the i th mined-versus-unmined comparison in the j th study (see equation (1)), which is normally distributed about an unknown true effect size θ_{ij} with sampling error e_{ij} , where $e_{ij} \sim N(0, v_{ij})$. The sampling variance v_{ij} is estimated from each comparison based on equation (2). The second level describes the prediction model for the true effect size θ_{ij} :

$$\theta_{ij} = \beta_0 + u_{ij} + w_j \quad (4)$$

where θ_{ij} is modelled by a mean effect size β_0 plus two random effects: u_{ij} , the random effect across comparisons (i) nested within studies (j) and w_j , the random effect of the j th study, which accounts for differences between studies. $u_{ij} \sim N(0, \sigma_{comp}^2)$, where σ_{comp}^2 is the among-comparison (within-study) variance and $w_j \sim N(0, \sigma_{study}^2)$, where σ_{study}^2 is the among-study variance. Combining the first- and second-level models (equations (3) and (4)), we get a hierarchical random-effects model:

$$L_{ij} = \beta_0 + u_{ij} + w_j + e_{ij} \quad (5)$$

which allows us to estimate the mean (overall) effect size based on observed effect sizes L_{ij} .

In the hierarchical mixed-effects model, the mean effect size of each factor (that is, aquatic invertebrates versus fish versus salamanders; reclaimed versus ongoing reclamation/active mining) of a categorical predictor variable (that is, taxon and mine-type) can be modelled by including each factor as an indicator predictor variable in the second-level model⁵³. From equation (4), the true effect size θ_{ij} of the i th comparison in the j th study can be expressed as:

$$\text{Taxa: } \theta_{ij} = \beta_{inv} X_{inv,ij} + \beta_{fish} X_{fish,ij} + \beta_{sala} X_{sala,ij} + u_{ij} + w_j \quad (6)$$

$$\text{Mine type: } \theta_{ij} = \beta_{rec} X_{rec,ij} + \beta_{act} X_{act,ij} + u_{ij} + w_j \quad (7)$$

where $\beta_{inv} \dots \beta_{act}$ are regression coefficients corresponding to the mean effect size of each factor where subscripts inv = aquatic invertebrates, rec = reclaimed, act = ongoing reclamation/active mining and so on, and $X_{inv,ij} \dots X_{act,ij}$ are indicator variables given a value of 1 if the i th comparison in the j th study corresponds to the factor, and 0 otherwise.

We performed this analysis using the `rma.mv` function in the `metafor` package⁵³ in R⁵². Confidence intervals were calculated based on the t -distribution akin to the Knapp-Hartung method⁵⁴ because of low sample sizes. Finally, we evaluated the potential for publication bias in our meta-analysis by graphical interpretation of funnel plots²⁵. The funnel plot for the richness effect size showed a negative-skew, largely due to ref.²⁰ (Supplementary Figs. 2,3). A plausible reason for the lower-than-expected effect size is that unmined sites are in larger watersheds than mined sites and that watershed size correlates positively with local macroinvertebrate richness⁵⁵. Excluding this study resulted in a symmetrical funnel plot and did not change estimated effect sizes substantially, suggesting that our findings are likely robust to publication bias. Data and code for our meta-analysis are available in Supplementary Data 1 and Supplementary Code.

Watersheds of conservation concern. To quantify watershed conservation-need in terms of endemism-weighted overall and T&E species richness, we used data on the native extant USGS HUC-8 watershed distribution of freshwater fish species from NatureServe²⁷ and federally T&E species from the US Fish and Wildlife Service (<https://www.fws.gov/endangered>). We manually delineated ranges of two T&E species that were considered as subspecies in the NatureServe dataset (*Etheostoma spilotum* as part of *E. sagitta*; *Oncorhynchus apache* as part of *O. gilae*) following on T&E species listing documents. We also revised the native range of *Catostomus santaanae* following determinations made in the T&E species listing. For two T&E species (*Emperichthys latos* and *Ictalurus pricei*) that were no longer found in their original native ranges, we included the transplanted range of each species because they were covered under the Endangered Species Act. Because the federally endangered *Etheostoma percnurum* is now considered a species complex

comprising four morphologically distinguishable species *E. percunum*, *E. lemniscatum*, *E. marmorpinnum*, and *E. sitikuense*³⁰, we regarded these four species as endangered and distinct in our analysis. We included only T&E species whose entire native range was listed as threatened or endangered (that is, species in which some populations were not protected, for example, various salmon species in the Pacific Northwest and various subspecies were excluded).

Following ref.²⁸, we incorporated endemism into the richness metric for a given location (watershed) by weighting species with smaller ranges (that is, higher endemism) more highly. The endemism-weighted total species richness ($\text{rich}_{\text{all},k}$) in the k th watershed was therefore:

$$\text{rich}_{\text{all},k} = \sum_{m=1}^M \frac{1}{W_m} \quad (8)$$

where W_m = the number of watersheds in which the m th species ($m = 1, 2, \dots, M$) occurring in watershed k is currently extant and native. Correspondingly, the endemism-weighted T&E richness ($\text{rich}_{\text{T\&E},i}$) in the i th watershed was:

$$\text{rich}_{\text{T\&E},i} = \sum_{q=1}^Q \frac{1}{W_q} \quad (9)$$

where W_q = the number of watersheds in which the q th threatened/endangered species ($q = 1, 2, \dots, Q$) occurring in watershed i is currently extant and native.

We estimated coal production across HUC-8 watersheds by distributing county-level coal production from 2005–2016³⁶ proportionally across overlapping watersheds based on potentially minable coal area³⁶. Potentially minable coal is defined as “coal of sufficient quality and energy content to justify extraction, based on existing data” according to the USGS National Atlas³⁷. We corrected a very small number of omissions and potential errors in the EIA county-level coal production dataset before using it for our analysis. Specifically, we corrected omissions and typographical errors in county and state names using information from the Mine Data Retrieval System (<https://arweb.msha.gov/drs/drshome.htm>) of the Mine Safety and Health Administration (MSHA). Entries without a valid MSHA mine ID were deleted because we could not track production of these mines across decadal periods. We also removed duplicated entries comprising multiple identical values of coal production reported by a given mine for a particular county in the same year, assuming they were erroneous. Invalid ($n = 751$) and duplicated entries ($n = 52$) accounted for 1.2% of all entries ($n = 69,250$) in the full 1983–2016 dataset. The removal of these entries was therefore unlikely to affect our results.

Given coal production for a given county, i , is P_i and that this county overlaps with N watersheds, coal production for the n th overlapping area (where $n = 1, 2, \dots, N$) that overlaps with this county $P_{i,n}$ was distributed proportionally to the area of potentially minable coal in the same area ($\text{MC}_{i,n}$):

$$P_{i,n} = \frac{\text{MC}_{i,n}}{\sum_n \text{MC}_{i,n}} \times P_i \quad (10)$$

For example, if a given county has an average annual production of 1 MMst and overlaps with two watersheds with potentially minable coal areas of 200 km² and 400 km² respectively, coal production in the two overlapping areas will be estimated as $200/600 \times 1 = 0.33$ MMst and $400/600 \times 1 = 0.67$ MMst respectively. We repeated this process across all counties in the contiguous US to calculate coal production across individual (county–watershed) overlapping areas; we did this using the Spatial Intersect and Zonal Statistics functions in ArcGIS 10.5³⁸. In one county (Franklin County, Alabama) that produced coal despite not appearing to overlap with potentially minable coal area³⁶, we distributed coal production in proportion to the area of overlapping watersheds. Finally, to calculate coal production across all HUC-8 watersheds, we summed coal production across all overlapping areas belonging to each watershed. To control for size differences across watersheds, we calculated coal production per unit watershed area as a metric of conservation-risk.

Watersheds of conservation concern (WCC) were defined as those ≥ 90 th percentile in coal production per unit potentially minable coal area³⁶ and the number of endemic (rich_{end}) or federally T&E aquatic species ($\text{rich}_{\text{T\&E}}$). We further defined two categories of WCC: the first (highest) category (WCC1) contains watersheds ranked in the top 5% in terms of coal production and endemism-weighted richness. The second category (WCC2) contains watersheds ranked in the top 10% in terms of coal production and endemism-weighted richness minus watersheds already placed in the first category.

The HUC-8 watershed scale was appropriate for identifying WCCs across the US because it was the smallest scale at which reliable data for freshwater fish distributions were available. Despite their being small enough to show spatial heterogeneity in coal production and richness patterns, we treated the HUC-8 watersheds as independent (that is, coal production within each watershed is the primary stressor; impacts from upstream watersheds do not substantially influence downstream watersheds). This assumption is reasonable because the size of an average watershed substantially exceeds the zone of impact from a mine in an upstream watershed. Using streamline data from the National Stream Internet³⁹, we found that the average HUC-8 watershed is drained by $\sim 2,403$ km of stream

(interquartile range: $\sim 1,271$ – $3,155$ km; values are estimates owing to minor differences between the USGS HUC-8 dataset used in our analysis⁴⁰ and the (older) version of USGS HUC-8 used in ref.³⁹), whereas the downstream impact of mining averages ~ 10 km (ref.³⁷); therefore, for a given watershed, the impact of mining from upstream watersheds is likely negligible compared to that within the watershed.

Our categorization of WCC1 (top 5% in both coal production and richness metrics) versus WCC2 (top 10%) versus non-WCC (other) watersheds was based on arbitrary thresholds. In addition, we did not incorporate potential differences between the effects of acid versus neutral–alkaline mine drainage and different methods of mining in different regions owing to the paucity of such data as explained in the main text. We therefore present our full results (Supplementary Data 2) including raw values of endemism-weighted richness and coal production metrics to allow readers to identify important watersheds based on their own thresholds and criteria, especially when studies comparing effects of region and mining methods become available.

Data availability. The datasets generated and analysed in our study (Supplementary Data 1,2; Supplementary Code) are included in this published article as Supplementary Information files.

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References

- US Energy Information Administration *Annual Energy Outlook 2017* (2017).
- Copeland, C. *The Office of Surface Mining's Proposed Stream Protection Rule: an Overview* (Congressional Research Service, 2015).
- Bernhardt, E. S. & Palmer, M. A. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. *Ann. NY Acad. Sci.* **1223**, 39–57 (2011).
- Palmer, M. A. et al. Mountaintop mining consequences. *Science* **327**, 148–149 (2010).
- The Effects of Mountaintop Mines and Valley Fills on Aquatic Ecosystems of the Central Appalachian Coalfields* EPA/600/R-09/138F (US Environmental Protection Agency, 2011).
- Bryant, G. & Childers, H. A *Survey of the Water Quality of Streams in the Primary Region of Mountaintop Mining/Valley Fill Coal Mining. Mountaintop Mining/Valley Fills in Appalachia Draft Programmatic Environmental Impact Statement, Appendix D* EPA 9-03-R-00013B (US Environmental Protection Agency, 2003).
- Lemly, A. D. *Selenium Assessment in Aquatic Ecosystems: A Guide for Hazard Evaluation and Water Quality Criteria* (Springer, New York, 2002).
- Hartman, K. J. et al. How much do valley fills influence headwater streams? *Hydrobiologia* **532**, 91–102 (2005).
- Kimmer, W. G. & Argent, D. G. Stream fish community responses to a gradient of specific conductance. *Water Air Soil Pollut.* **206**, 49–56 (2010).
- Griffith, M. B., Norton, S. B., Alexander, L. C., Pollard, A. L. & LeDuc, S. D. The effects of mountaintop mines and valley fills on the physicochemical quality of stream ecosystems in the Central Appalachians: a review. *Sci. Total Environ.* **417–418**, 1–12 (2012).
- Lindberg, T. T. et al. Cumulative impacts of mountaintop mining on an Appalachian watershed. *Proc. Natl Acad. Sci. USA* **108**, 20929–20934 (2011).
- Opinion and Order. People of the State of Illinois V. Freeman United Coal Mining Company, Llc, A Delaware Limited Liability Company, and Springfield Coal Company, Llc, A Delaware Limited Liability Company; Environmental Law and Policy Center V Freeman United Coal Mining Company, Llc, A Delaware Limited Liability Company, and Springfield Coal Company, Llc, A Delaware Limited Liability Company (Consolidated: Pcb 10-61 and 11-2)* (Illinois Pollution Control Board, 2012).
- Pond, G. J. Biodiversity loss in Appalachian headwater streams (Kentucky, USA): Plecoptera and Trichoptera communities. *Hydrobiologia* **679**, 97–117 (2012).
- Pond, G. J. et al. Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. *J. North Am. Benthol. Soc.* **27**, 717–737 (2008).
- Howard, H. S. et al. *Kentucky Mountaintop Mining Benthic Macroinvertebrate Survey: Central Appalachian Ecoregion, Kentucky. Mountaintop Mining/Valley Fills in Appalachia Draft Programmatic Environmental Impact Statement, Appendix D* EPA 9-03-R-00013B (US Environmental Protection Agency, 2003).
- Merricks, T. C. et al. Coal-mine hollow fill and settling pond influences on headwater streams in southern West Virginia, USA. *Environ. Monit. Assess.* **129**, 359–378 (2007).
- Armstead, M. Y., Yeager-Seagle, J. L. & Emerson, L. Benthic macroinvertebrate studies conducted in mountaintop mining/valley fill influenced streams in conjunction with the USEPA environmental impact study. *Proc. Am. Soc. Min. Reclam.* **2004**, 87–93 (2004).

18. Pond, G. J. et al. Long-term impacts on macroinvertebrates downstream of reclaimed mountaintop mining valley fills in Central Appalachia. *Environ. Manage.* **54**, 919–933 (2014).
19. Muncy, B. L., Price, S. J., Bonner, S. J. & Barton, C. D. Mountaintop removal mining reduces stream salamander occupancy and richness in southeastern Kentucky. *Biol. Conserv.* **180**, 115–121 (2014).
20. Fritz, K. M. et al. Structural and functional characteristics of natural and constructed channels draining a reclaimed mountaintop removal and valley fill coal mine. *J. North Am. Benthol. Soc.* **29**, 673–689 (2010).
21. Ferreri, C. P., Stauffer, J. R. & Stecko, T. D. Evaluating impacts of mountain top removal/valley fill coal mining on stream fish populations. *Proc. Am. Soc. Min. Reclam.* **2004**, 576–592 (2004).
22. Stauffer, J. R. Jr & Ferreri, C. P. *Characterization of Stream Fish Assemblages in Selected Regions of Mountain Top Removal/valley Fill Coal Mining. Mountaintop Mining/Valley Fills in Appalachia Draft Programmatic Environmental Impact Statement, Appendix D* EPA 9-03-R-00013B (US Environmental Protection Agency, 2003).
23. Wood, P. B. & Williams, J. M. Impact of valley fills on streamside salamanders in Southern West Virginia. *J. Herpetol.* **47**, 119–125 (2013).
24. Chao, A. & Jost, L. Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. *Ecology* **93**, 2533–2547 (2012).
25. Sutton, A. J. in *The Handbook of Research Synthesis and Meta-Analysis* 2nd edn (eds Cooper H., Hedges, L. V. & Valentine, J. C.) 435–452 (Russell Sage Foundation, New York, 2009).
26. *Annual Production Data from Annual Coal Report 2016* (US Energy Information Administration, 2017).
27. Digital distribution of native US fishes by watershed v.3.0. *NatureServe* <http://www.natureserve.org/conservation-tools/data-maps-tools/digital-distribution-native-us-fishes-watershed> (2010).
28. Kier, G. et al. A global assessment of endemism and species richness across island and mainland regions. *Proc. Natl Acad. Sci. USA* **106**, 9322–9327 (2009).
29. *Crystallaria cincotta*: IUCN Red List of Threatened Species <https://doi.org/10.2305/IUCN.UK.2013-1.RLTS.T202435A15362696.en> (2013).
30. Blanton, R. E. & Jenkins, R. E. Three new darter species of the *Etheostoma pecunurum* species complex (Percidae, subgenus *Catonotus*) from the Tennessee and Cumberland river drainages. *Zootaxa* **1963**, 1–24 (2008).
31. *Etheostoma spilotum*: IUCN Red List of Threatened Species. <https://doi.org/10.2305/IUCN.UK.2014-3.RLTS.T18243637A19033189.en> (2014).
32. *Chrosomus Cumberlandensis*: IUCN Red List of Threatened Species. <https://doi.org/10.2305/IUCN.UK.2014-3.RLTS.T17064A15364473.en> (2014).
33. Merriam, E. R., Petty, J. T., Strager, M. P., Maxwell, A. E. & Ziemkiewicz, P. F. Complex contaminant mixtures in multistressor Appalachian riverscapes. *Environ. Toxicol. Chem.* **34**, 2603–2610 (2015).
34. van der Welle, M. E. W., Roelofs, J. G. M. & Lamers, L. P. M. Multi-level effects of sulphur-iron interactions in freshwater wetlands in the Netherlands. *Sci. Tot. Environ.* **40**, 426–429 (2008).
35. Armstead, M. Y., Bitzer-Creathers, L. & Wilson, M. The effects of elevated specific conductivity on the chronic toxicity of mining influenced streams using *Ceriodaphnia dubia*. *PLoS ONE* **11**, e0165683 (2016).
36. Rose, A. W. & Cravotta, C. A. III *Geochemistry of Coal Mine Drainage. Coal Mine Drainage Prediction and Pollution Prevention in Pennsylvania* (Pennsylvania Department of Environmental Protection, Harrisburg, 1998).
37. Petty, J. T. et al. Landscape indicators and thresholds of stream ecological impairment in an intensively mined Appalachian watershed. *J. N. Am. Benthol. Soc.* **29**, 1292–1309 (2010).
38. Schorr, M. S. & Backer, J. C. Localized effects of coal mine drainage on fish assemblages in a Cumberland Plateau stream in Tennessee. *J. Freshwat. Ecol.* **21**, 17–24 (2006).
39. Smucker, N. J. & Vis, M. L. Use of diatoms to assess agricultural and coal mining impacts on streams and a multiscale case study. *J. N. Am. Benthol. Soc.* **28**, 659–675 (2009).
40. Schmidt, T. S., Soucek, D. J. & Cherry, D. S. Integrative assessment of benthic macroinvertebrate community impairment from metal-contaminated waters in tributaries of the Upper Powell River, Virginia, USA. *Environ. Toxicol. Chem.* **21**, 2233–2241 (2002).
41. MacCausland, A. & McTammany, M. E. The impact of episodic coal mine drainage pollution on benthic macroinvertebrates in streams in the Anthracite region of Pennsylvania. *Environ. Pollut.* **149**, 216–227 (2007).
42. Bott, T. L. et al. Abandoned coal mine drainage and its remediation: impacts on stream ecosystem structure and function. *Ecol. Appl.* **22**, 2144–2163 (2012).
43. Palmer, M. A. & Hondula, K. L. Restoration as mitigation: analysis of stream mitigation for coal mining impacts in southern Appalachia. *Environ. Sci. Tech.* **48**, 10552–10560 (2014).
44. Osenberg, C. W., Shima, J. S., Miller, S. L. & Stier, A. C. in *Marine Protected Areas: A Multidisciplinary Approach* (ed. Claudet, J.) 143–167 (Cambridge Univ. Press, 2011).
45. *Stream Protection Rule (Dec 20 2016)* 81 FR 93066 (US Federal Govt, 2016).
46. West Virginia to improve oversight of coal-mining pollution. Federal review of West Virginia coal mining reveals chronic failures *Center for Biological Diversity* https://www.biologicaldiversity.org/news/press_releases/2017/coal-02-07-2017.php (2017).
47. Bernhardt, E. S. et al. How many mountains can we mine? Assessing the regional degradation of central Appalachian rivers by surface coal mining. *Environ. Sci. Tech.* **46**, 8115–8122 (2012).
48. Hendryx, M. & Ahern, M. M. Mortality in Appalachian coal mining regions: the value of statistical life lost. *Public Health Rep.* **124**, 541–550 (2010).
49. *2012 National Survey of Hunters and Anglers* (Chesapeake Beach Consulting, 2012).
50. Hedges, L. V., Gurevitch, J. & Curtis, P. S. The meta-analysis of response ratios in experimental ecology. *Ecology* **80**, 115–1156 (1999).
51. Hsieh, T. C., Ma, K. H. & Chao, A. An R package for rarefaction and extrapolation of species diversity (Hill numbers). *Methods Ecol. Evol.* **7**, 1451–1456 (2016).
52. R Core Team R: *A Language and Environment for Statistic Computing* (R Foundation for Statistical Computing, 2017).
53. Viechtbauer, W. Conducting meta-analyses in R with the metafor package. *J. Stat. Softw.* **36**, 1–48 (2010).
54. Knapp, G. & Hartung, J. Improved tests for a random effects meta-regression with a single covariate. *Stat. Med.* **22**, 2693–2710 (2003).
55. Jacobsen, D. Low oxygen pressure as a driving factor for the altitudinal decline in taxon richness of stream macroinvertebrates. *Oecologia* **154**, 795–807 (2008).
56. *US Coal Fields* <https://hifld-geoplatform.opendata.arcgis.com/datasets?q=Coal%20Fields%20of%20the%20United%20States> (US Geological Survey, 2001).
57. *Regulatory Impact Analysis of the Stream Protection Rule* (Office of Surface Mining Reclamation and Enforcement, Industrial Economics, 2016).
58. ArcGIS Desktop 10.5 (ESRI, 2016).
59. Nagel, D. et al. *National Stream Internet Hydrography Network for Spatial-Stream-Network (SSN) Analysis: Rocky Mountain Research Station* https://www.fs.fed.us/rm/boise/AWAE/projects/NationalStreamInternet/NSI_network.html (US Forest Service Data Archive, 2017).
60. *Watershed Boundary Dataset and Lines for HUC2-12* (US Geological Survey, 2014).

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Author contributions

X.G. conceived the study, performed the analyses, and led the writing of the paper. J.D.O. and D.S. provided input into data analyses and contributed to writing the paper.

Competing interests

The authors declare no competing interests.

Additional information

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