

Geomorphic degradation and heavy metal contamination associated with artisanal alluvial tin and aggregate mining in a Rwandan headwater river

Aelis Spiller

aelis.spiller@mail.mcgill.ca

McGill University

Mette Bendixen

McGill University

Placide Habinema

Rwanda Mines, Gas and Petroleum Board

Jean Pierre Bizimana

University of Rwanda

Christian von Sperber

McGill University

Lars Lonsmann Iversen

McGill University

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Abstract

Mining is a cornerstone of economic development across Sub-Saharan Africa, yet the environmental impacts of artisanal and small-scale mining remain poorly understood, particularly for commodities other than gold. Rwanda exemplifies the region's rapid economic transformation, where tin, tungsten, and tantalum (3Ts) constitute major national exports, and quarrying provides construction-grade aggregates for rapidly expanding infrastructure in the country. We evaluated how artisanal alluvial tin and aggregate mining affects the geomorphology, sediment dynamics, and heavy metal contamination in the Rusine River watershed in the Rutongo Mining Concession, Rwanda's largest producer of tin-concentrate. Using measurements from 55 sites along a 19km reach of the Rusine River and its tributaries, we assessed turbidity, riverbank erosion, channel morphology, and sediment heavy metal concentrations. We show that riverbank erosion risk was augmented at active mining sites, and turbidities were elevated throughout the main channel, indicating catchment-scale mining-induced sedimentation. Arsenic and chromium concentrations in alluvial sediments were elevated throughout the watershed to levels harmful to aquatic life. These results demonstrate that artisanal alluvial mining degrades both the physical structure of the watershed and quality of benthic sediments. While the most severe impacts occur at active mining sites, effects extend throughout the catchment via downstream sediment transport and contaminant mobilization. As mining and infrastructure development accelerate across Sub-Saharan Africa, our study provides an important example of the environmental challenges that can accompany this trajectory.

Introduction

Mining is a foundational material economy of the modern world, unpinning the development of infrastructure, energy production, and technology. Since the year 2000, global mineral extraction has grown by over 50% and the push for technological advancement and the transition to green energy will continue to intensify demand for precious metals (Kirsanov & Saaya, 2024). Population growth and urbanization, concentrated in low- and middle-income countries, are also amplifying the demand for local and regional construction materials (Zhong et al., 2022). In fact, sand and gravel (aggregates) make up the most extracted group of materials worldwide, even exceeding fossil fuels (Krausmann et al., 2009).

Despite its foundation to sustainable development, mining causes adverse environmental impacts, influencing up to 50 million square kilometers of Earth's land surface (Sonter et al., 2020). The annual production of 7–9 billion tonnes of solid mine wastes makes up one third of the global sediment budget (Macklin et al., 2023; Winde & Newman-Portela, 2026). Rivers are important recipients and vectors of these mine wastes and pollutants (Sonter et al., 2018). Due to the lack, or failing, of tailings dams, mining wastes are deposited and stored in river channels, by which they are transported throughout the mine drainage network.

The primary concerns of mining pollution in rivers are acid mine drainage, heavy metal contamination, and chemical pollution via erosion and sedimentation (Mardonova & Han, 2023). Mining exposes buried geological materials, augmenting the former two processes (Wilde & Newman-Portela, 2026) which may occur naturally to lesser extents via acid rock drainage and geogenic heavy metal occurrence (Singh et al., 2003; Somers & McKenzie, 2020). Low order mining-affected streams are particularly vulnerable to extreme contamination, as they have not been diluted by non-mining sources (Macklin et al., 2023).

Surface and groundwaters are also at risk of being modified in both terrestrial and alluvial mining operations. Rivers are diverted via dams and channel realignment and dewatering reduces base flows (Harding & Boothryd, 2005; Winde & Newman-Portela, 2026). Terrestrial mining can also affect river systems indirectly via land cover change. Mining results in deforestation, and more broadly devegetation. This land conversion modifies the sediment budgets of rivers within the catchment, alters flow regimes, and erosion rates may result in eutrophication downstream (Bruijnzeel, 2004). Compacted soil limits infiltration and increases runoff, leading to flashier flows and more extreme seasonal swings.

Although much of the volume of extracted minerals is concentrated in few countries, including China, the United States, and Australia (Kirsanov & Saaya, 2024), many low-and middle-income countries are economically reliant on the mining sector (International Council on Mining and Minerals, 2025). Regions such as Sub-Saharan Africa and Southeast Asia are emerging as hotspots of new, poorly regulated mining operations (Macklin et al., 2023). Much of the mining that occurs in these regions is artisanal and small-scale mining (ASM) due to non-existing or less (enforced) regulation around mining and quarrying, high labor supplies, and poverty-driven informal economies (Franks, 2020).

There is no universally-agreed upon definition of ASM. ASM is characterized as being low-tech and labor-intensive (Hilson & Maconachie, 2017) and typically targets more superficial deposits than large-scale mining, such as alluvial and eluvial deposits as well as discarded tailings (World Bank Group, 2019). ASM is critical to local livelihoods and poverty alleviation. Globally, the sector provides direct

employment to 40.5 million people, supporting the livelihoods of 130–270 million people (Girard et al., 2024; World Bank Group, 2019). ASM can be up to five times more profitable than local agriculture industries (World Bank Group, 2019), and thus people supplement or replace their income from subsistence agriculture. For example, artisanal tin miners in Rwanda can earn up to \$17 USD a day (Macháček & Dušková, 2016), while the poverty line, under which nearly 30% of the population falls, is at just over \$1 USD/day (National Institute of Statistics Rwanda, 2024). Most ASM operations occur in rural, agricultural areas. This may foment conflict between miners and farmers as informal ASM can encroach into and degrade agricultural lands in addition to polluting key water sources (Ofosu et al., 2020).

ASM is especially prevalent in Sub-Saharan Africa (SSA), where it is the second highest employer after the agricultural sector (Ofosu et al., 2020). Here, ASM is poverty-driven; with subsistence agriculture becoming a less viable livelihood in SSA due to rising living costs, land degradation, and climate change, ASM is an increasingly important element of the livelihood portfolio (Macháček et al., 2022). Despite its growing economic importance in SSA, ASM continues to occupy a peripheral position in the economic development agenda of the region, allowing artisanal mining operations to remain largely informal, environmentally degrading, and with poor working conditions (Hilson & McQuilken, 2014).

Much of the global and African literature on ASM focuses on gold mining (Marrufo et al., 2025). This leaves a dramatic knowledge gap in the effects of the extraction of other valuable minerals and aggregates (Bendixen et al., 2023). This is of particular importance, as Africa is the continent with the highest projected demand for aggregates within the next 100 years (Zhong et al., 2022) due to population growth and urbanization. The combination of increasing demand and limited knowledge on current extraction activities in SSA means that aggregates, like metals, are extracted using illegal and unsustainable methods with human and environmental consequences.

Here, we study how artisanal alluvial mining in the Rusine River in central Rwanda is modifying and polluting the water course. We analyze the physical and chemical effects of alluvial tin and aggregate mining. Across a gradient of land use, we measure river sedimentation, erosion risks, and heavy metal contamination. We predict that alluvial mining will have acute localized effects on erosion risk and heavy metal contamination, with sedimentation persisting downstream of mining operations. We further assess tributaries unaffected by mining to determine any background impacts from the industrial mining waste in the area and compare that to the signals of watershed degradation induced specifically by artisanal alluvial mining activities.

Methods

Study site

The Rusine River watershed is located in the southeastern portion of the Rulindo District in the Northern Province of Rwanda (Fig. 1). It encompasses an area of approximately 65km² and extends from 1.732° to 1.81°S latitude and 29.99° to 30.103°E longitude. A second order stream, the Rusine makes up part of the headwaters of the Nile River Basin. The outlet of the Rusine River into the Nyabugogo River is less than 25km from the capital city of Kigali.

Rwanda is a small, landlocked nation in the Great Lakes Region of East Africa. It has the highest population density in continental Africa (Bendixen et al., 2023) and is undergoing rapid urbanization under the government's "Vision 2050" plan. Mining and quarrying are pillars of Rwanda's economic development (Rwanda Mines, Petroleum, and Gas Board, 2025). The country is positioning itself as a major player in the production and trade of the 3Ts – tin, tantalum, and tungsten – exporting \$1.1 billion (USD) of minerals in 2023 (Rwanda Development Board, n.d.). Rwanda is among the largest global producers of each of the 3Ts (U.S. Geological Survey, 2023). Simultaneously, like across SSA, there is an exploding demand for local construction materials to meet the infrastructure needs of a developing, industrializing, urbanizing country (Zhong et al., 2022).

In Rwanda, 90% of rural areas are fed by springs and groundwaters (Global Water Leadership Programme, 2022), rendering the impacts of alluvial mining on the quality and integrity of freshwaters are especially important. Located in the Rutongo Mining Concession, The Rusine River is one such rural mining-affected stream. The Rwandan Ministry of Environment (2018) has identified the Rusine River as particularly polluted with sediments from mining and quarrying. In the Rutongo Mining Concession, commercial-scale mines target underground tin ores. High concentrations of arsenic, chromium, and cobalt have been measured in waste materials from Rutongo Mines (Ishimwe et al., 2023).

The Rutongo Mining Concession is located in the Kibara orogen, a large metallogenic province in Central Africa characterized by many granite-related ore deposits (Pohl, 1994). This area has a legacy of tin extraction, dating back to 1931. Belgian mineral companies primarily exploited the alluvial deposits for the first decades of extraction. Now, commercial operations only target primary quartz vein mineral occurrences at six operational underground tin mines within the concession. (Dewaele et al., 2010). Rutongo Mines is not only the

largest tin concentrate producer, but also the largest private sector employer, in Rwanda (Ishimwe et al., 2023; Samuel & Alexandre, 2022). Alluvial tin deposits and riverbed aggregates are exploited by artisanal miners. Rivers, such as the Rusine, are hotspots of artisanal mining operations in the Rutongo Mining Concession, and consequently, much of the conversion from forested to bare land has occurred along river streams (Kazindu et al., 2020). Scattered rural communities throughout the region engage in smallholder agriculture, livestock rearing, and forest management for domestic energy and timber production.

Data Collection:

We sampled 55 sites along an approximately 19km stretch of the Rusine River and its tributaries in June and July of 2024. In June 2025 we revisited the watershed to collect riverbed sediments to test for heavy metal contamination, collecting sediment from 11 sites. We classified each of the original 55 sites as either Mining (M), Non-mining (Nm), or Tributary (T) (Fig. 1). We hereon refer to these categories as site types. All sites with mining, aggregate and/or tin, main channel and tributary, are classified as M. Those without mining are further subdivided by their position in the main channel (Nm) or tributaries (T). Ultimately, we sampled 23 M sites, 25 Nm sites, and seven T sites.

Characterization of hydrological and geomorphological variables

At each site, the turbidity of the river water was measured three times in situ with a YSI ProDSS multimeter. Discharge was measured with the mid-section method (Turnipseed & Sauer, 2010) where the width of the wet channel was divided into five sections and the depth of the midpoint of each section was noted. With an Owen's hydropropeller and a stopwatch, the flow velocity was calculated in each section, positioning the hydropropeller to sit at ~ 60% of the depth of the water column from the water surface (40% of the flow depth from the bed). For sites where the depth of the river was too low to take velocity measurements at five intervals, the discharge was calculated in fewer sections (one to four).

We report channel depth as the maximum depth recorded in the discharge profile of each site. An Emlid Reach 2 + dGPS was used to construct stream gradient profiles and cross sections of the river profile, including the riverbanks. Raw data was processed with Emlid Flow software. In order to calculate the stream gradient of the entire stretch of river sampled, the distance from start of sampling was calculated with the measure distance tool in Google Maps.

Riverbed substrate was scooped with a trowel into a 26 x 30cm tray for the purposes of categorizing substrate by dominant grain size per a modified Udden-Wentworth classification (Table S1). We recategorize "very coarse gravel" in the original classification as "pebbles" and silt is the finest grain size we identify to. The tray containing the substrate and a metric ruler for post-hoc classification was photographed, allowing us to classify the dominant grain size visually. To minimize the bias introduced by making this classification based on individual expert opinion, three individual contributors classified and compared the dominant substrate at ten sites, coming to a joint conclusion for consistency.

To measure erosion risk, we adapted Rosgen's Bank Erosion Hazard Index (BEHI) (Table S2) (Rosgen, 2001). We lack reliable bankfull metrics, but characterized the root depth to bank height ratio, root density, surface protection, and bank angle of each site in the field. We confirmed the bank angle with the dGPS data. We multiplied the index thresholds by 0.8 given that we dropped the variable of the bank height to bankfull height ratio.

Heavy metals

We used a makeshift sediment grab sampler to remove the top layer of riverbed sediments, taking triplicate samples per site. We prioritized collecting the finer grains for ease of later processing. In the laboratory, the sediment was oven dried for 48 hours at 60°C. Once devoid of moisture, we ground each individual sample with a mortar and pestle to a powder.

We measured an array of elemental concentrations, including arsenic, chromium, lead, and iron (Table S3) in the sediment samples using a Bruker S1 Titan Model 800 portable XRF analyzer (Bruker Nano Analytics, Germany). We loaded approximately 1–2 g of sample into a 1.5 cm diameter XRF sample cup, leveled it to a height of 0.6 cm and covered it with a taut Chemplex® prolene thin-film SpectroMembrane® secured by a plastic ring. Sample cups were filled to a consistent height (0.6 cm) to ensure uniform measurement geometry, maintain constant sample thickness and packing, keep a fixed distance to the analyzer window, and reduce edge effects (Mantler and Schreiner, 2000). For each site, we filled three sample cups with separate subsets of the total sediment sample and measured each triplicate three times with the analyzer, ultimately producing nine readings per site.

Statistical analysis

We ran linear Gaussian contrast and marginal models with post-hoc tests to determine the influence of land use, watershed position, and/or substrate on the continuous, numerical variables we measured: turbidity (Formazin Nephelometric Units (FNU)), arsenic (ppm), chromium (ppm), lead (ppm), BEHI, channel width (m), channel depth (m), flow velocity (m/s), and stream gradient (m/m). All models were corrected for contributing upstream drainage area (CUDA) to account for the position in the watershed, thus removing the potential influence of the natural gradients in the river-continuum on our evaluation of the effects of land use and substrate. We obtained flow accumulation values at 3 arcsecond spatial resolution from the HydroSHEDS database (Lehner et al., 2011), matching the coordinates of our sites to raster values. For coordinates not identified as being in a water course, we identified the closest appropriate cell and assigned its flow accumulation value to the site. We converted flow accumulation cells to CUDA (km²) by multiplying cell value by 0.00859, the approximate conversion of 3 arcseconds to square kilometers at the latitudinal range of the Rusine River (~ 1.8°S). To better meet the assumptions of the model, we log transformed the arsenic concentrations.

We performed a Chi² test to test for the overall association between land use and substrate. Finally, to compare stream gradients between main channel sites based on mining presence, we utilized a post-hoc t test, adjusting p-values for multiple comparisons using the single-step adjustments. All statistical analyses were performed in RStudio 4.3.1 using the *stats* (v4.3.1; R Core Team, 2023) and *multcomp* (v1.4.28; Hothorn et al., 2025) packages (Spiller et al., 2026).

Results

Mining activities

Alluvial artisanal mining activities take place throughout the Rusine River continuum and more broadly throughout the catchment area. However, more intense mining efforts, of both tin and aggregates, become more pronounced downstream, beginning at, (-1.76818, 30.06255), approximately 11km from the start of sampling. Tin mining operations are characterized by digging pits along the periphery of the wet channel or directly into the river bank. Then, the ore-bearing rock is washed in the running water of the river. The process is described in detail by Macháček & Dušková (2016).

Aggregate miners remove stones, sand, and gravel from the wet channel and depositional features. In the most downstream stretch of the Rusine River, in-stream sand mining occurs in the absence of tin extraction. Here, the riverbanks have been reinforced with chopped bamboo to minimize erosion and sustain mining activities (Fig. S1a).

Artisanal alluvial mining in the Rusine River is transient and opportunistic. There is evidence of abandoned tin mining pits throughout the water course. We identified active mining sites in all riverbed substrate types; there is no correlation between mining and substrate ($\chi^2(4, N = 55) = 3.6, p = 0.5$). To that point, although in-stream sand mining targets finer substrates (more downstream), there are operations to remove sediment as big as cobbles from the channel to feed the numerous stone crushing operations along the river.

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Active channel characteristics

The stream gradient across the sampled 19.155km is -13.5m/km or 1.35%. This gradient changes throughout the sampled stretch of river, decreasing linearly in magnitude by 0.8m/km². This corresponds to a change in slope from roughly 3% to 0.3% across the sampled stretch of the Rusine River. These gradients are characteristic of plane bed and pool-riffle systems (Montgomery & Buffington, 2013). The observed decrease in stream gradient is expected by the natural river continuum, as stream gradient decreases with increasing catchment area (Lewin et al., 2018).

Localized stream gradients in the main channel are lower at Mining sites than at Non-mining sites ($t(34) = -2, p < 0.05$). However, when accounting for CUDA, mining becomes an insignificant driver of stream gradient within the main channel (Nm-M: mean gradient difference = 0.00255m/m, $p = 0.43$). Given that mining activities increase downstream, it is difficult to detangle the natural and anthropogenic modifiers of stream gradient.

Both the wet channel width ($F(1,53) = 36.1, p < 0.001$) and depth ($F(1,53) = 42.9, p < 0.001$) increase roughly linearly with increasing CUDA, as expected by the hydraulic geometry. We find that mining does not modify the wet channel width ($F(2, 51) = 2.6587, p = 0.08$) nor depth

($F(2, 51) = 0.3801$, $p = 0.69$), even when accounting for CUDA.

Flow regime

The Rusine River itself is a perennial stream, although some of its tributaries are ephemeral. Over the roughly 19km sampled, the dry season discharge increased more than four-fold, from $0.07\text{m}^3/\text{s}$ to $0.33\text{m}^3/\text{s}$ at its outlet, directly upstream of a dam preceding its confluence with the Nyabugogo River. There is no trend in velocity with increasing CUDA ($F(1,53) = 2.346$, $p = 0.132$). This is contrary to the expectation of a slight increase in velocity in the upstream to downstream direction given that velocity is proportional to flow depth per Manning's equation (Leopold & Maddock, 1953).

Substrate and sediment

The Rusine River retains characteristics of a natural river, where the riverbed in substantial upstream stretches is dominated by cobbles, pebbles, and gravel but the stream eventually transitions to sandy and silty substrates (Doretto et al., 2020). Similarly, as expected, the turbidity increases with increasing CUDA ($F(1,42) = 5.5$, $p < 0.05$) (Fig. 2).

Turbidity is elevated throughout the main channel of the Rusine River in comparison to its tributaries that are unaffected by mining (Fig. 3a). The difference is significant, even accounting for CUDA (Post hoc differences between T-M: mean turbidity difference = -850.308 FNU, $p = 0.006$ and T-Nm: mean turbidity difference = -855.532 , $p = 0.003$). The turbidity of waters in tributaries with no mining is on par with that of the main channel (Mining tributaries: mean turbidity = 1212.3 FNU and main channel: mean turbidity = 1206.3 FNU), suggesting that the presence of mining has catchment-wide consequences in the downstream direction. Furthermore, localized turbidities within the main channel do not differ between Mining and Non-mining sites (mean difference = 5.224 FNU, $p = 0.999$).

Some mining-derived sediment does settle shortly downstream. We observe that within the main channel of the Rusine River, downstream distance from mining and CUDA collectively explains observed variation in turbidity ($F(2, 41) = 3.721$, $p = 0.033$). Turbidity tends to decline with increased downstream distance from mining ($\beta = -65.27$, $p = 0.185$), though this phenomenon is counteracted by the increase in turbidity with increasing CUDA ($\beta = 0.063$, $p = 0.146$), and when considered together, neither effect is significant.

Within the main channel, local turbidity is significantly correlated with riverbed substrate, independent of CUDA ($F(5.5,5) = 38$, $p < 0.01$). When including tributary sites, however, the significant association between turbidity and riverbed substrate disappears ($F = 1.26$, $p = 0.298$). Generally, turbidity increases with decreasing grain size, although gravel beds are associated with lower turbidity than pebble beds (Fig. 3b). This suggests two things: 1) with the transition from coarser to finer substrates in the downstream direction, local sediment suspension, in addition to the accumulation of suspended sediment from upstream, contributes to the increasing trend in turbidity with CUDA, and 2) the presence of mining is such a strong driver of increased turbidity that the signal of turbidity contributed by substrate is insignificant within mined channels.

Erosion

Mining drives riverbank erosion in the Rusine River. The BEHI at Mining sites was significantly higher than that at Non-mining and Tributary sites (Post hoc differences between M-Nm: mean BEHI difference = 10.53 , $p < 0.001$ and M-T: mean BEHI difference = 13.5 , $p < 0.001$) (Fig. 4a). The BEHI at Non-mining and Tributary sites was comparable (mean BEHI difference = 2.973 , $p = 0.4$). Perhaps more concerning than the *comparatively* elevated erosion risk at Mining sites, is the *objectively* high erosion risk at Mining sites, as classified by the BEHI thresholds. Nearly all Mining sites have high erosion risk, while that at Non-mining and Tributary sites hover between moderate and low risk. High BEHI sites are characterized by (near) vertical, devegetated banks, often with toe scour and evidence of past mass wasting.

The color of the water in the main channel of the Rusine River and in its tributaries affected by mining is a deep orange-brown (Fig. 4b). This is indicative of the erosion of lateritic soils from the riverbank and with subsurface sediments unearthed directly by mining. This is consistent with the high iron concentrations measured in the riverbed sediments, which are assumed to be in part derived from these highly weathered soils (Table S3). We also observe that the sinuosity of the Rusine River increases in mining-affected stretches (Fig. S3). The enhanced erosion and collapse of riverbanks increase lateral migration of the channel and meander growth. The sinuosity "recovers" at non-mining stretches downstream of mining.

Heavy metal contamination

Arsenic (As) concentrations throughout the riverbed sediments of the Rusine River are elevated to levels considered to be of high contamination that pose a risk to aquatic life ((New York State Department of Environmental Conservation [NYSDEC], 2014)) (Fig. 5). The

highest concentrations of As in sediments were measured in tributaries of the Rusine River with mining (T_m) present. These concentrations were significantly higher than those in tributaries without mining (T_{Nm}) ($T_m - T_{Nm}$: mean $\log(\text{As})$ difference = 0.96, $p < 0.01$). There is no significant difference in As between main channel and non-mining tributary sediments, but this is likely a result of there being only two T_{Nm} sites (Type II error). Indeed, As concentrations are higher in all main channel sites than T_{Nm} sites.

All of the riverbed sediments sampled were moderately or highly contaminated with Chromium (Cr) (Fig. 5) (NYS DEC, 2014). The concentration of Cr in the main channel of the Rusine River decreased downstream (Fig. S2). Additionally, more upstream tributaries, regardless of the presence of alluvial mining, had higher Cr concentrations. This suggests there is some dilution of Cr in the downstream direction, with the source(s) of Cr either located or more concentrated higher upstream in the watershed. Unlike arsenic, it seems that artisanal alluvial mining does not further mobilize Cr. Some of the sampled sites were also moderately to highly contaminated with lead (Pb) (Fig. 5) (NYS DEC, 2014). The highest concentrations of Pb were measured in T_m sites, but there is otherwise no consistent pattern across site type or longitudinally.

Discussion

Mining activities in the Rutongo Concession

Although there is no evidence of tin artefacts or tin mining in Rwanda prior to the 20th century, archeological evidence indicates that some of the earliest iron production in SSA occurred in Rwanda (Humphris, 2012). This legacy of smelting natural ores may have equipped Rwandans to engage in artisanal tin extraction upon the discovery of cassiterite deposits by German and Belgian surveyors (Behuria, 2026). Communities in the Rutongo Mine Concession have likely been extracting alluvial aggregates since prehistoric times and they have been familiar with tin mining since at least 1931, nearly 100 years ago.

In the past century, the economy and livelihoods of communities in the Rutongo Mine Concession have incorporated artisanal tin extraction, with waves of industrial extraction during the colonial period and now post-genocide. After the exit of colonial era operators, there were decades of primarily informal artisanal mining in the region (Mambwe et al., 2026). Since 2015, Rutongo Mines has been credited with transforming this largely artisanal operation into a model of organized, formalized, modern mining (Mambwe et al., 2026). Trinity Metals (2025), the parent company of Rutongo Mines, claims that they employ 2500 locals, making up 98% of their workforce. However, Samuel & Alexandre (2022) argue that many of the workers are migrants and that suboptimal recruitment of local miners increases illegal mining in the concession.

Regardless, it is undeniable that locals continue to engage in artisanal tin mining, particularly in the Rusine River and some of its tributaries, and that Rutongo Mines is concerned about this unauthorized mining. In their 2025 Annual Due Diligence Report, Rutongo Mines documented that 119 of the 135 reported incidents in 2025 were related to the legality/legitimacy of mining operations. They classify unauthorized mining as a high risk.

Artisanal alluvial mining occurs, and has occurred, all throughout the Rusine River (Fig. 1; Fig. S1). The location, intensity, and type of these mining activities are influenced by geology, access to roads, and other land uses. Stretches of the Rusine River that are currently unaffected by mining and show no signs of previous mining are in turn protected by geology, accessibility, and land use. For example, at site 55 (6.35km downstream from the most upstream sampling point, Fig. 1) 1) there are boulders throughout the channel, 2) the channel is far from a road or path, and 3) the land beyond both sides of the riverbanks is steep and covered in eucalyptus trees that will be harvested to make charcoal. It would be difficult to remove aggregate material from the wet channel and bank toe due to boulder armoring and to transport that material away from the river. Furthermore, the eucalyptus stands are too valuable as a source of domestic energy to remove. These stretches are important local refuges for biodiversity. Mining does occur in tandem with agricultural and grazing areas. These land uses don't impede access to the river and have more dynamic, flexible boundaries than stands of maturing trees.

As a consequence of the spatial position of artisanal mining in and near rivers, ASM causes particularly severe impacts to soil and water via the pollution and diversion of water courses (Boldy et al., 2021). The environmental impacts of large-scale mining tend to be better monitored, regulated, and managed than those of ASM. So, while larger operations have a greater potential for impact, they also have a greater capacity to minimize damage (Sonter et al., 2018). For example, although Cano-Lodoño et al. (2023) found that open-pit gold mining in Colombia has a magnitude higher environmental impact than artisanal alluvial gold mining, informal artisanal operations may use mercury and other banned toxic substances. Because much of ASM occurs informally (at times also illegally), these operations are not subject to official environmental impact assessments and do not meet the operational guidelines expected of large-scale mining (Tarra et al., 2022).

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Artisanal alluvial tin and aggregate mining both contribute to the geomorphic changes and the erosion of riverbanks and sedimentation of the Rusine River. However, because of the distinct extraction methods, alluvial tin and aggregate mining may also have different geomorphological consequences. Tin mining carves out/into riverbanks, resulting in toe scour, bank collapse, and widening of the active floodplain. Tin mining disrupts the continuity of stream profiles by redirecting and damming the water course, disturbing soil and drainage systems (Hodder, 1959). Aggregate mining in the wet channel directly resuspends fine grains, elevating the suspended sediment concentration with alluvial sediments. Aggregate mining also results in riverbank erosion via headcutting and the hungry water effect (Kondolf, 1997).

According to the literature, both forms of alluvial mining are expected to increase the channel width (Damseth et al., 2024; Knighton, 1989). However, in the Rusine River channel width is unaffected. Likewise, alluvial tin mining tends to cause bed aggradation by accumulation of terrestrial sediments and mining waste, while aggregate mining lowers the bed (Islam et al., 2024; Knighton, 1989). However, channel depth in the Rusine River is also unaffected by mining. We speculate that at the smaller scale of ASM operations, wet channel modifications are less pronounced compared to larger-scale alluvial mining. We note though, that these data only consider the river during dry season with a relatively meager discharge. With higher volumes of water, we would expect differences between the wet channel at Mining and Non-mining sites to be more pronounced due to the modification of the active floodplain, riverbank collapse, channel aggradation, and/or channel incision. Additionally, it may be that in a low order stream like the Rusine that has been somewhat channelized, alluvial mining operations are more transformative to the active floodplain than the wet channel itself (Iversen et al., 2025).

We observe that artisanal alluvial mining in the Rusine River has local effects on erosion risk. BEHI is elevated at Mining sites, while Non-mining and Tributary sites are stable. Both Non-mining and Tributary sites share the same land use pressures along the riverbanks, namely agriculture and occasionally some natural (shrubs) or planted (bamboo) riparian vegetation. However, we do note that the BEHI of Mining sites surrounded by agriculture is higher than that of Mining sites in forested stretches. There is a compounded effect of land use, with the limited erosion control capacity of crop root systems leaving riverbanks more vulnerable to erosion from mining. Unlike riverbank erosion, mining activities elevate the suspended sediment load throughout the whole watershed. Thus, the presence of alluvial mining has catchment wide consequences in the downstream direction. Not all adverse mining impacts can be contained to the mining locale.

Deviations from the natural river continuum

Alluvial mining in the Rusine River may accelerate the transition of the channel from a pool riffle stream to a plane bed stream lacking the expected heterogeneity (Montgomery & Buffington, 2013). Some stretches even resembling dune-ripple channels, perhaps in part because of the removal of coarse substrates for stone crushing and gravel production. Mining activities have completely taken over and modified the active floodplain of some stretches of the Rusine, artificially creating braided streams via the construction of secondary channels for ore washing and piling of aggregates throughout the floodplain. Mining activities induce more lateral erosion, increasing sinuosity. This is a localized impact, with non-mining stretches downstream of mining activities returning to a less-meandering pattern.

Mining also modifies the sediment dynamics in the watershed. Although the gradient, sinuosity, and width-depth ratio of the Rusine River are consistent with bed load and mixed load channels (Schumm, 1985), the % bed load is attenuated due to the high suspended sediment concentration. The turbidity of the Rusine River waters even exceeds that of its trunk stream, the Nyabugugo River, by 10 to 30-fold.

Consequently, the zones of sediment production and deposition are altered. We speculate that in this case both zones expand and overlap. Riverbank erosion is still rampant in downstream stretches otherwise expected to have gentler bank slopes and shear stresses (Townsend, 1980). This input of sediment contributes to downstream, and even local, bed aggradation. Removal of aggregates from the river channel may counteract said aggradation (Islam et al., 2024).

The unnatural suspended sediment loads and accelerated transition away from coarse substrates is consequential for the biological communities throughout the Rusine River watershed. In the Rusine River, coarser substrates support higher biomass and biodiversity in macroinvertebrate communities. Primary productivity is stifled by high turbidities (Bilotta & Brazier, 2008). The consequences transfer to secondary production and are amplified by clogging filterers' feeding structures (Newcombe & MacDonald, 1991). Egg survival and fry emergence of fish is also generally negatively impacted by elevated suspended sediment (Chapman et al., 2014).

Heavy metal contamination

We measure concentrations of As and Cr throughout the Rusine River watershed that are classified by the NYS DEC as being unsafe for aquatic life. While this research shows that alluvial mining increases As concentrations, it is not possible from the data obtained here to

determine the exact contribution of difference sources to the heavy metal-laden riverbed sediments. Rutongo Mines identifies emissions/waste to be of high risk in their 2025 Annual Due Diligence Report. Rutongo mine tailings, which have elevated As and Cr concentrations, are disposed of as small dumps covering approximately 72.5 hectares in more than 13 different places in the Rusine River catchment (Ishimwe et al., 2023). A number of streams in the watershed traverse these waste rock dumps. Although we know that alluvial mining operations contribute to heavy metal accumulation, it is likely that these contributions are further exacerbated by background heavy metal concentrations caused by the runoff and leaching of Rutongo mine tailings. Historic alluvial mining wastes also may contribute to elevated heavy metal concentrations.

Chromium

Geogenic Cr is associated with ultramafic rocks, not granites like those of the Kibara orogen (Chrychoou et al., 2016). Cr concentrations in both the riverbed sediments and tailings are enriched above granitic background levels, as we measured 50-310ppm of Cr in Rusine riverbed sediments and Ishimwe et al. (2023) measured 8.6-128ppm in Rutongo mine tailings. As such, while some Cr in the riverbed sediments may be derived from Rutongo mine tailings, alluvial mining also elevates Chromium levels. We speculate that much of the modern Cr concentrations originate from historical alluvial mine wastes in the water course that over time have adsorbed and concentrated Cr over time. These are likely located more upstream given the dilution of sediment Cr concentration in the downstream direction. An analysis of the chromium stable isotope composition may be able to distinguish between different origins, but this was beyond the scope of our study. Chromium exhibits toxicity to aquatic plants and animals and is carcinogenic to humans (Pushkar et al., 2021).

Arsenic

Arsenopyrite, the most abundant arsenic mineral, is associated with tin ores in high-temperature hydrothermal veins (Ng et al., 2022). The presence of abundant tin ores, as evidenced by the underground and alluvial tin mining activities in the watershed, indicates that there is geogenic As in the underlying geology of the Rusine River watershed.

This raises the question, are the concentrations of As measured in the sediments of the Non-mining Tributaries (T_{Nm}) sites due to natural exposure of As-bearing geologies, or are mining activities elevating the baseline of mobile arsenic in the entire catchment by leaching As into groundwater that ultimately feeds tributaries and the baseflow of the Rusine? We observe that alluvial mining does increase As in the riverbed sediments of the Rusine watershed. This is believed to occur through the exposure of As-rich subsurface material via riverbank erosion as well as the resuspension of fine particles that bind to dissolved As. In a case study of alluvial tin mining in Rwanda's Western Province, Macháček & Dušková (2016) identify the main culprit of elevated As concentrations in the river water to be chemical leaching during the sluicing of mined ore. The same practice occurs in the Rusine River. Particulates from the mined material settle into the riverbed during the washing process.

Additionally, fine alluvial particulates adsorb aquatic arsenic species. Iron oxides and clay minerals especially are significant arsenic adsorbers (Stollenwerk, 2003). As major components of the lateritic soils whose erosion is contributing to the suspended sediment load in the Rusine River, we expect there to be significant adsorption of dissolved arsenic by suspended sediment. A portion of this arsenic-bearing sediment will be transported downstream, while some will settle locally at the interface of the riverbed and water column, making up the habitat of benthic organisms.

The high contamination of arsenic in the Rusine River watershed poses risk to aquatic communities and public health. Inorganic arsenic, like that in arsenopyrite, accumulates in phytoplankton and algae and is transferred through the food web (Foust Jr. et al., 2016; Reuther, 1992). Macroinvertebrates uptake arsenic by filtering and ingesting particulate matter. High As levels have been shown to decrease the density, biomass, and biodiversity of macroinvertebrate communities (Chi et al., 2017).

In fish, arsenic accumulates in internal organs, particularly the liver and kidneys (Kumari et al., 2016). Arsenic enters fish's bodies directly from the water column as well as via their diet. Diet-borne arsenic exposure may be the primary route of bioaccumulation in fish (Zhang et al., 2011), with the accumulated As concentrations in primary producers (Reuther, 1992) and macroinvertebrates (Spehar et al., 1980) well exceeding that of the water column. High arsenic concentrations can stunt fish growth, be deleterious to internal organs, and cause death (Kumari et al. 2016).

The World Health Organization sets 10 µg/L (0.01ppm) as the provisional guideline for arsenic in drinking water (WHO, 2017). Smith & Smith (2004) suggest that it is reasonable to elevate the drinking water standard to 50 µg/L in developing countries with widespread arsenic exposure in drinking water. While we did not measure the dissolved arsenic concentrations in the Rusine River watershed, we measured the arsenic in the riverbed sediments to range from 26–455ppm. This is 3 to 4 orders of magnitude higher than even the more

lenient 50 µg/L standard. Consequences for prolonged As ingestion include skin, lung, and bladder cancers (Smith & Smith, 2004). Local communities do not utilize the Rusine River for drinking water, likely due to the elevated turbidity. They rely on surface water in the non-mining tributaries and shallow-dug wells. However, these sources likely still have unsafe As concentrations.

Local communities' exposure to arsenic may be yet amplified through the accumulation of As in crops and foodwebs. Arsenic in irrigation water will accumulate in the topsoil, ultimately becoming available for uptake by roots. Although different plants uptake arsenic more readily than others and preferentially store arsenic in different parts of the plant – e.g., roots, leaves, stem, grain – an increase in arsenic concentration in agricultural soils increases arsenic concentrations throughout the plant (Heikens et al., 2007; Rosas-Castor et al., 2016). This study does not attempt to confirm nor quantify arsenic accumulation in crops grown throughout the Rusine River watershed, but we highlight the potential risk caused by the stunning arsenic contamination we identify.

Conclusion

Although important to local livelihoods, artisanal alluvial mining contributes to the degradation of the integrity of the Rusine River. All mining activities cause riverbank erosion and sedimentation, altering the geomorphology and sediment budget of the watershed. Tin mining also causes heavy metal pollution, with arsenic and chromium of primary concern to aquatic life and local populations that rely on the Rusine River and its tributaries for domestic water and irrigation.

This study contributes to the growing body of literature on the environmental impacts of alluvial mining. Whereas most previous studies have focused on large scale sand mining in major rivers, particularly in Southeast Asia (Koehnken et al., 2019), our work highlights the consequences of artisanal alluvial aggregate and tin mining in a Sub-Saharan African headwater system. This study contributes to the representation of Sub-Saharan Africa, artisanal mining, headwaters, and coarser grain extraction to the corpus of alluvial aggregate mining.

Given the paucity of scientific literature addressing the environmental impacts of alluvial tin mining from the 21st century our findings from the Rusine River watershed exemplifies the continued relevance of the (artisanal) exploitation of alluvial tin deposits and that these activities continue to represent an important but under-recognized challenge to freshwater ecosystems.

As mining intensifies in response to population growth and economic development, adopting strategies to minimize environmental impact, while continuing to offer societal benefits, is critical to the sustainable provisioning of ecosystem services and the protection of Rwanda's valuable biodiversity. Formalizing artisanal mining activities could facilitate improved environmental oversight and promote the adoption of practices that reduce erosion, sedimentation, and heavy metal pollution.

Declarations

Competing interests:

No, I declare that the authors have no competing interests as defined by Springer, or other interests that might be perceived to influence the results and/or discussion reported in this paper.

Dual publication

No, the results/data/figures in this manuscript have not been published elsewhere, nor are they under consideration by another publisher.

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Author Contribution

Conceptualization: Aelis Spiller (A.S.), Lars Lonsmann Iversen (L.L.I.), Mette Bendixen (M.B.), Placide Habinema (P.H.), Jean Pierre Bizimana (J.P.B.); Data curation: A.S.; Formal Analysis: A.S., L.L.I.; Funding Acquisition: L.L.I., M.B., J.P.B.; Investigation: A.S., L.L.I., M.B., P.H.; Methodology: A.S., L.L.I., M.B., P.H., Christian von Sperber (C.V.S.); Project Administration: A.S., L.L.I., M.B.; Resources: A.S., L.L.I., M.B.; Software: A.S., L.L.I.; Supervision: L.L.I.; Validation: A.S., L.L.I.; Visualization: A.S., L.L.I.; Writing Original: A.S.; Writing Revisions: A.S., L.L.I., M.B., P.H., J.P.B., C.V.S

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Data Availability

All of the data used in this study was generated by these authors for this study. This data is publicly available in a Zenodo repository at: <https://zenodo.org/records/21569362>

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81. Supplemental:

Figures

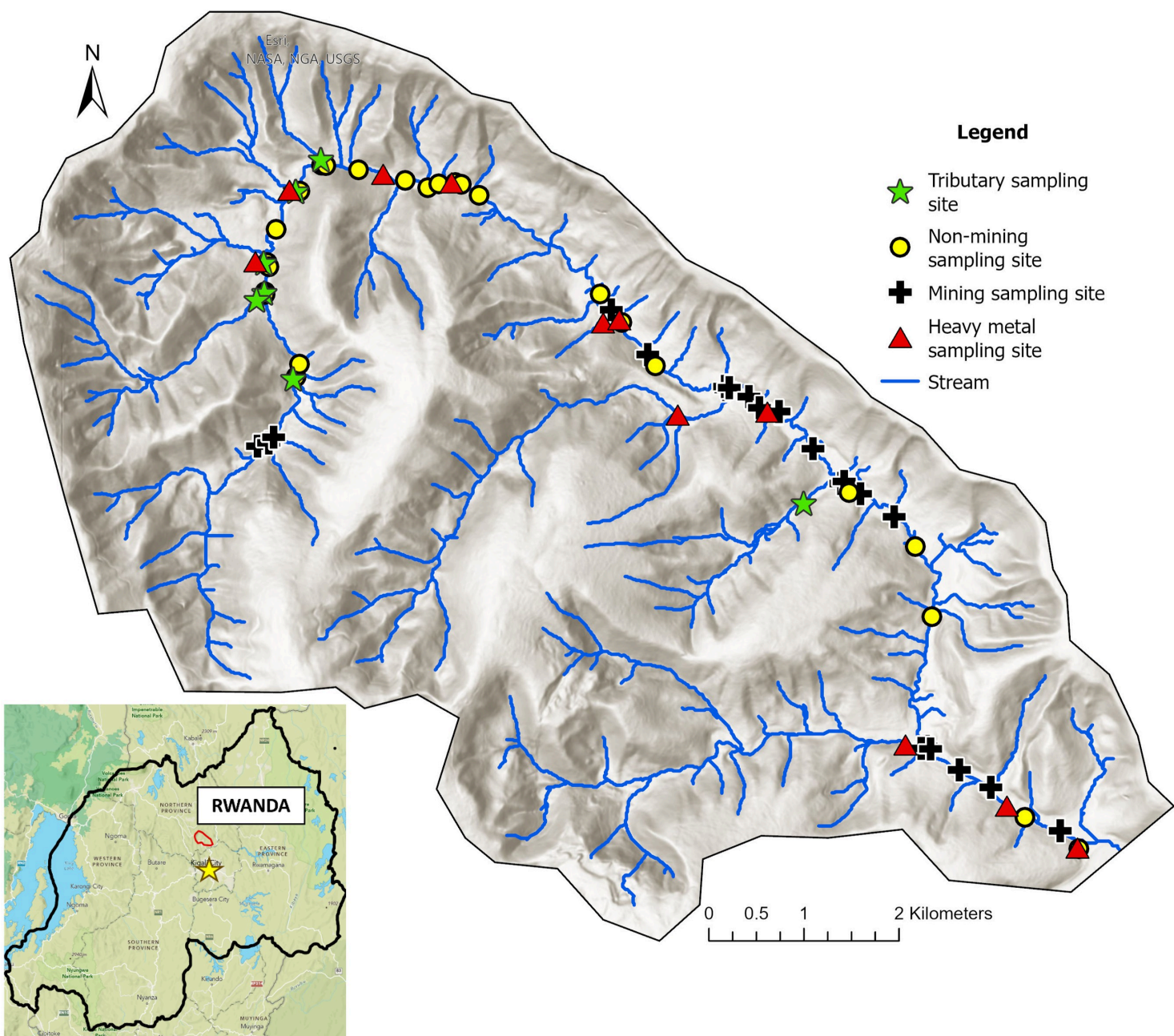


Figure 1

Map of Rusine watershed and sampling locations in central Rwanda. The red outline on the inset represents the Rusine River watershed and the yellow star the capital city of Kigali.

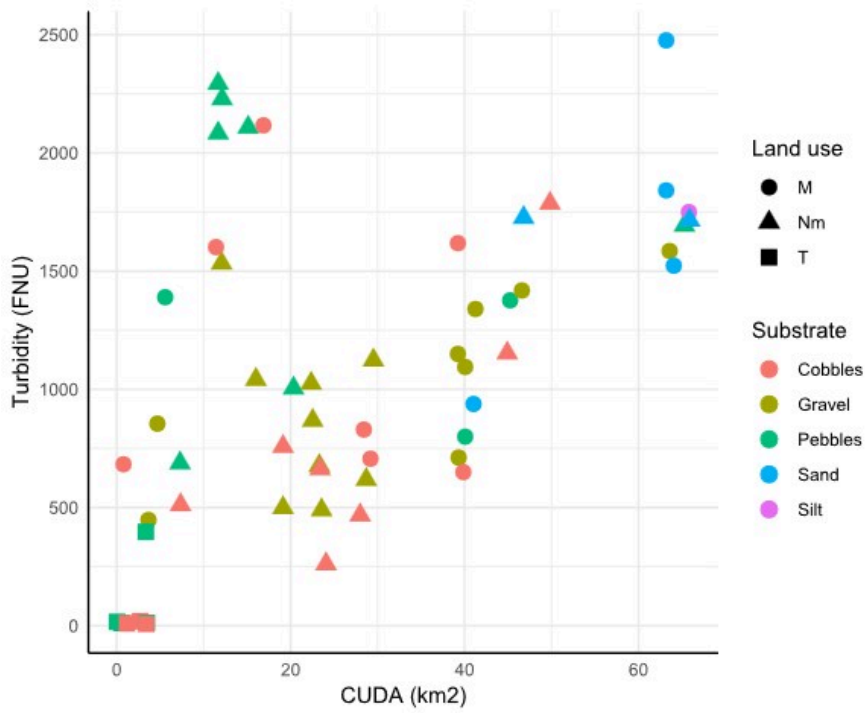


Figure 2

Turbidity along a gradient of contributing upstream drainage area with the dominant substrate and land use defined for all 55 sites.

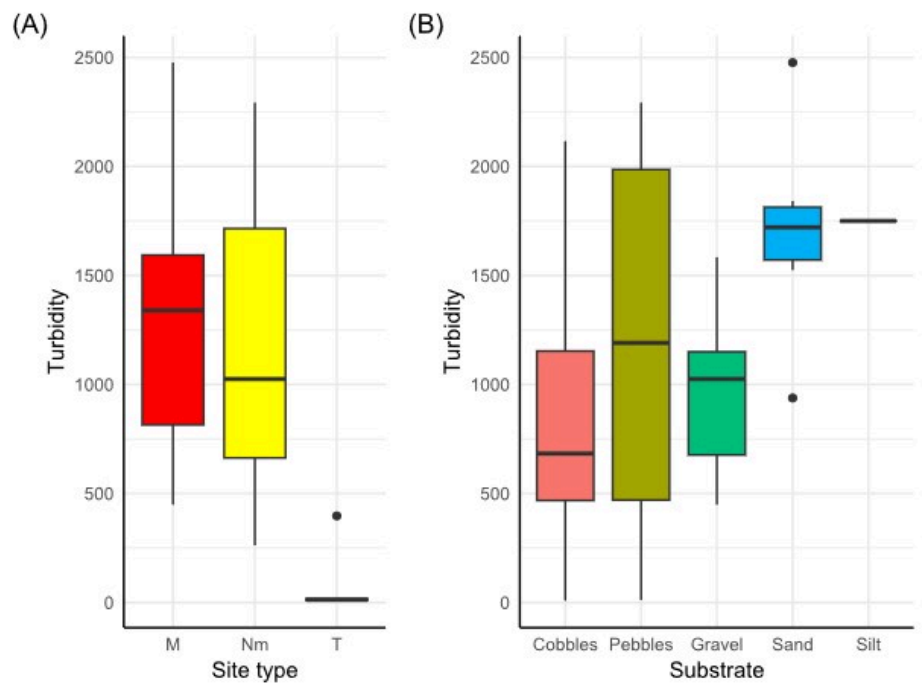


Figure 3

Turbidity of the Rusine River by a) site type and b) dominant substrate. The central line indicates the median and the box widths the interquartile range.

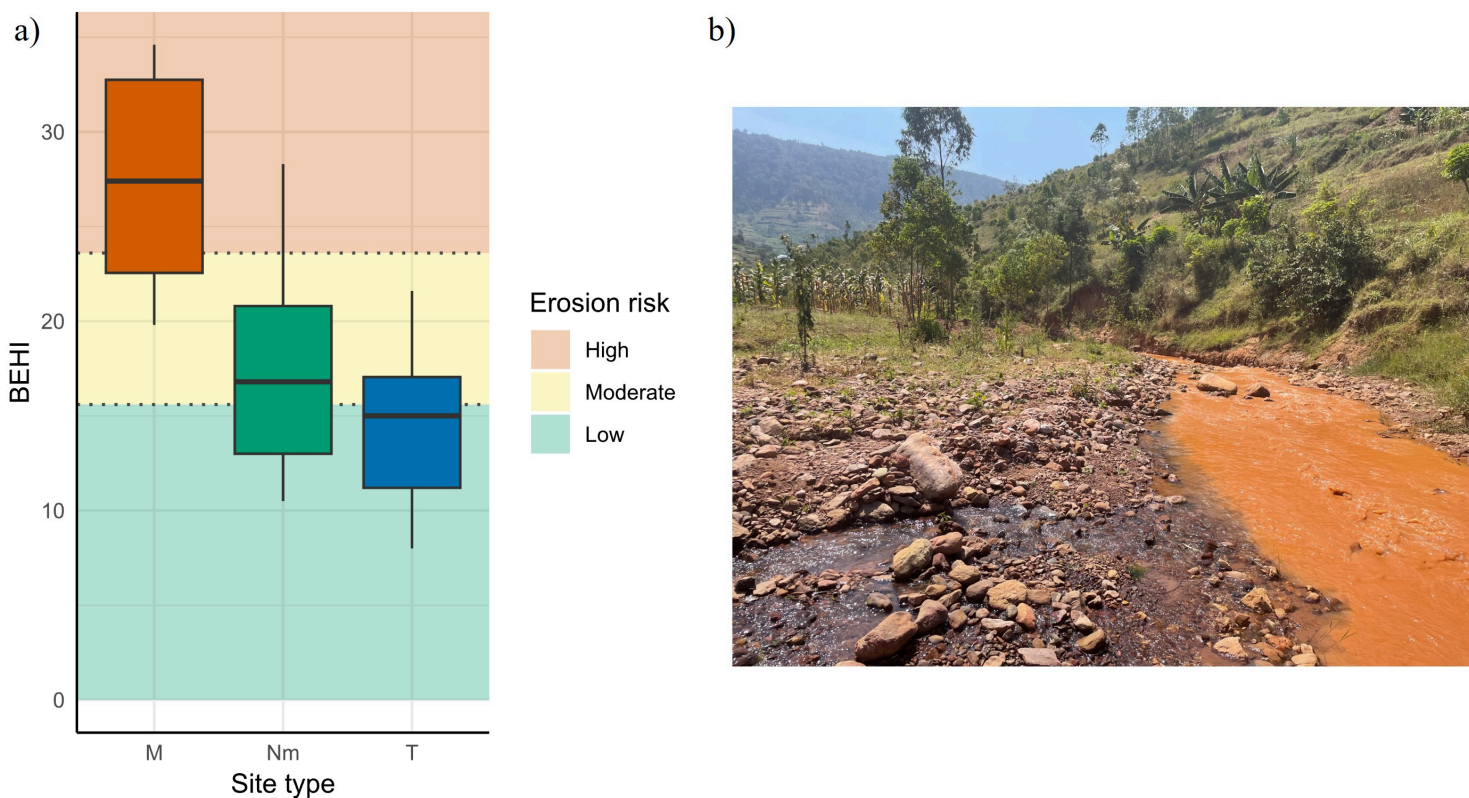


Figure 4

a) BEHI of Rusine River banks by site type with erosion risk categories; the central line is the median and the box widths the interquartile range, b) confluence of a tributary (left in photo) with the Rusine River (reddish stream to the right in photo), demonstrating the stark difference in turbidity.

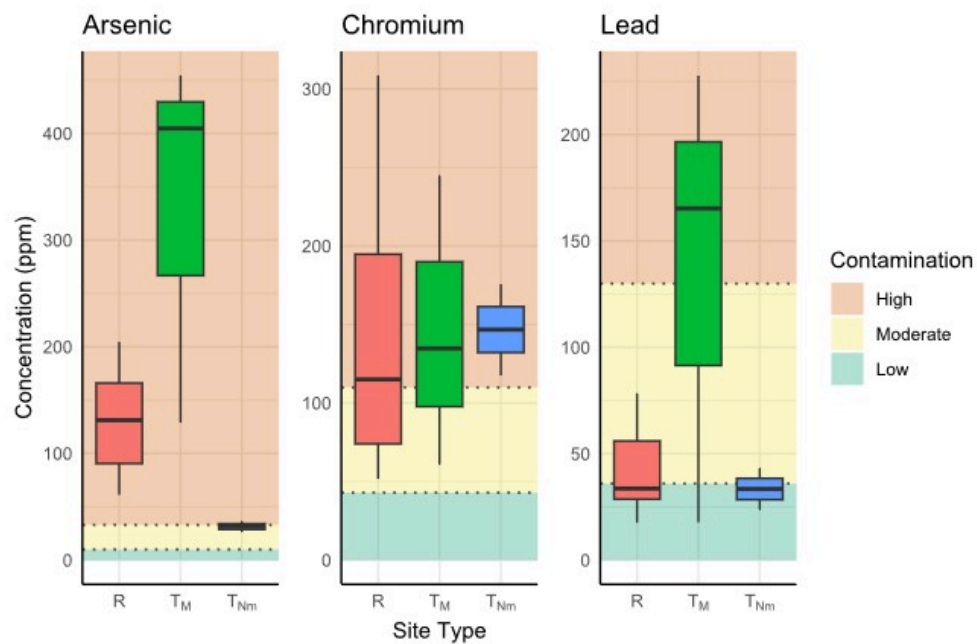


Figure 5

Heavy metal concentrations (arsenic, chromium, lead) of riverbed sediments by (modified) site type – Rusine River, mining tributary, and non-mining tributary. Contamination levels defined by the NYS DEC are shown as zones of from high, moderate, and low. The central line is the median and the box width the interquartile range.

Supplementary Files

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