Published in final edited form as:

Environ Sci Technol. 2018 May 15; 52(10): 5851–5858. doi:10.1021/acs.est.7b05730.

Phytoremediation Reduces Dust Emissions from Metal(loid)-Contaminated Mine Tailings

Juliana Gil-Loaiza^a, Jason P. Field^b, Scott A. White^a, Janae Csavina^c, Omar Felix^d, Eric A. Betterton^c, A. Eduardo Sáez^d, Raina M. Maier^{a,*}

^aUniversity of Arizona, Department of Soil- Water and Environmental Science, Tucson, AZ 85721, USA

^bUniversity of Arizona, School of Natural Resources and the Environment, Tucson- AZ 85721, USA

^cUniversity of Arizona, Department of Hydrology and Atmospheric Sciences, Tucson- AZ 85721, USA.

^dUniversity of Arizona, Department of Chemical and Environmental Engineering, Tucson- AZ 85721, USA.

Abstract

Environmental and health risk concerns relating to airborne particles from mining operations have focused primarily on smelting activities. Yet there are only three active copper smelters and less than a dozen smelters for other metals compared to an estimated 500,000 abandoned and un-reclaimed hard rock mine tailings in the US, that have the potential to generate dust. The problem can also extend to modern tailings impoundments, which may take decades to build and remaining barren for the duration, before subsequent reclamation. We examined the impact of vegetation cover and irrigation on dust emissions and metal(loid) transport from mine tailings during a phytoremediation field trial at the Iron King Mine and Humboldt Smelter Superfund (IKMHSS) site. Measurements of horizontal dust flux following phytoremediation reveals that vegetated plots with 16% and 32% canopy cover enabled an average dust deposition of 371.7 and 606.1 g m⁻² y⁻¹, respectively, in comparison to the control treatment which emitted dust at an average rate of 2,323 g m⁻² y⁻¹. Horizontal dust flux and dust emissions from the vegetated field plots are comparable to emission rates in undisturbed grasslands. Further, phytoremediation was effective at reducing the concentration of fine particulates, including PM₁, PM_{2.5}, and PM₄, which represent the airborne particulates with the greatest health risks and the greatest potential for long-distance

*Corresponding author: Raina M. Maier, Department of Soil, Water, and Environmental Science, 1177 E. 4th Street, University of Arizona, Tucson, AZ 85721-0038, rmaier@ag.arizona.edu, Phone: 520-621-7231, Fax: 520-626-6782.

Present Addresses

Janae Csavina is now at the National Ecological Observatory Network- NEON, Calibration- Validation and Audit Laboratory, Boulder, CO 80301, USA.

Omar Félix Villar is now at SRK consulting, Tucson- AZ 85741, USA.

Supporting Information

This material is available free of charge via the Internet at <http://pubs.acs.org>.

Additional information about the area of study, field sampling, instrument placement, and design, particle sizes generated in a temporal scale, horizontal dust flux data per seasonal sampling with their respective wind roses, metal(loid)s mass collected, and exponential functions used to calculate average total horizontal dust flux can be found in this section.

transport. This study demonstrates that phytoremediation can substantially decrease dust emissions as well as the transport of windblown contaminants from mine tailings.

Graphical Abstract

1. Introduction

Hard rock mining activities are prevalent in arid and semiarid regions throughout the world. Associated ore-processing activities like grinding, smelting, and refining as well as waste containment facilities such as mine tailings and waste rock dumps can be potential sources of atmospheric dust and airborne contaminants that may affect surrounding communities and ecosystems^{1,2}. While ore-processing activities cease when mining activities are completed, the mine tailings that remain represent a potential long-term dust source that may persist for years to centuries, depending on tailings characteristics (Fig. 1). In general, mine tailings have poor soil structure³⁻⁵, and low organic matter and nutrient content⁶. Legacy tailings may also have extreme pH and high metal(loid) or radionuclide content⁷. As a result, such sites often do not support adequate vegetation cover. This leaves the soil surface exposed to the erosion leading to the transport, dispersion, and deposition of tailings particles and associated hazardous contaminants into neighboring soils and water sources⁸⁻¹², including areas that may be important for agriculture¹³⁻¹⁵. This is intensified in arid and semiarid environments because of extreme high temperatures and low precipitation^{16,17}. Recent studies suggest that wind dispersion is one of the most important routes of exposure for communities close to mine tailings^{8,9,16,18-21}.

Mine tailings particles can range from colloidal to coarse particle size^{2,5,6,16}. Both particle size and wind speed play an important role in particle and contaminant dispersion^{22,23}. Coarse particles (60–2000 μm) are transported primarily by saltation and typically account for the majority of mass movement at the local scale²⁴. Saltation is the process by which coarse particles bounce along the soil surface causing detachment of finer particles and producing an avalanching effect of increasing dust emission with increasing distance downwind of an eroding surface²⁵⁻²⁷. This mechanism is largely responsible for the

detachment of silt- and clay-sized particles ($< 60 \mu\text{m}$) from the surface²⁵ while finer particles can be carried by wind to regional distances^{12,25,28}. Environmental factors that reduce dust generation include the presence of physical and biological barriers like rocks, plant litter, soil crust, and vegetation^{25,26}. The type, cover, and distribution of vegetation is an important factor influencing the ability of the wind to reach the soil surface and thus controls, to a large degree, the rate of saltation²⁹. In general, even unvegetated surfaces immediately downwind of vegetation (within 5-10 times the height of the plant) are relatively protected from the erosive force of the wind by the plant^{26,30,31}. Revegetation is currently being explored as a potential long-term solution to the stabilization of mine tailings³²⁻³⁴. As this technology is explored, it is important to quantify its potential for reducing dust emission and contaminant transport from mine tailings. The objective of this study is to evaluate the effectiveness of a phytoremediation field trial³⁴ on reducing contaminant transport and dust emissions from metal(loid)-contaminated mine tailings.

2. Materials and methods

2.1 Site Description

The study was conducted at the Iron King Mine and Humboldt Smelter Superfund (IKMHSS) site in Dewey-Humboldt (population ca 4,000), Arizona (North $34^{\circ}31'57''$, West $112^{\circ}15'9''$). The site was placed on the National Priority List in September 2008 because of high levels of metal(loid)s associated with the tailings, mainly arsenic and lead³⁵. Characteristics of the 62-ha mine tailings pile include extreme acidity (pH 2.5 to 3.5), low organic matter and nitrogen content (below detection limits), low numbers of neutrophilic heterotrophic bacteria $145 \pm 2 \text{ CFU g}^{-1}$ dry soil, and high levels of As ($2,590 \text{ mg kg}^{-1}$) and Pb ($2,200 \text{ mg kg}^{-1}$)³⁴. The main tailing pile is 33.5 m high above the original surface elevation occupying a total area of $96,000 \text{ m}^2$, and $4,893,000 \text{ m}^3$ in volume³⁵. The surficial tailings (0-30 cm) have a loam texture with a particle size distribution of 20.4% clay, 44.8% silt, and 34.7% sand^{5,36}.

The IKMHSS area has a topography prone to steep slopes and a vegetation characterized as “Interior Chaparral Shrubland” which is comprised of grasslands, and evergreen shrublands on a patchy mosaic pattern on a soil low in nutrients^{34,37,38}. The study area is semiarid and has a mean annual rainfall of approximately 310 mm y^{-1} , which is distributed primarily during the summer (June – Aug.) and winter (Dec. – Feb.) months³⁴. The spring and fall months are typically characterized by low atmospheric humidity, low soil moisture, and intermittent periods with strong wind gusts^{8,39}. Meteorological data was obtained with an onsite wireless Vantage Pro™ 2 Plus weather station (Davis Instruments Corp, Hayward, CA), located in the center of the study area to monitor weather conditions including air temperature, relative humidity, wind speed and direction, rainfall, and soil temperature (Table S1). In general, wind speed at the study site is greatest in the morning with frequent wind gusts throughout the day and typically decreasing in the evening³⁹. The predominant wind directions at the site are southerly, southwesterly, and southeasterly (Fig. S1; Fig. S2)³⁹.

2.2 Study Design

A phytoremediation field trial was initiated in May 2010 on the main tailings pile consisting of 24 irrigated study plots (six treatments replicated in quadruplicate covering an area of 0.27 ha) to evaluate the impact of compost amendment (0, 10, 15, 20% w/w) on revegetation success³⁴. A randomized block design was used except for the 0% compost controls, which were located at the far corners of the study area to avoid contamination with compost. Compost was incorporated into the top 20 cm of the tailings followed by seeding with a mix of six native plants. Supplemental irrigation (358 mm y⁻¹) was applied following seeding. Seedlings successfully germinated and established resulting in a plant cover dominated by buffalo grass (*Buchloe dactyloides*) and quailbush (*Atriplex lentiformis*).

In the present study, horizontal dust flux was measured on four study plots at the site from May 2011 to May 2012, each plot was approximately 9.6 m wide by 15 m long. Three plots were from the study area described above and the fourth plot, which received no irrigation or compost was adjacent to the study area. The plots were carefully chosen to meet two criteria. For the two vegetated plots, the first criterion was that they represent only the 15% compost amendment level. The second criterion, which was also applied to the control plots, was that each plot was similarly influenced by wind (e.g., plots in the center of the field trial were not used).

For the two vegetated plots used in this study, canopy cover was determined using transect and quadrat sampling at the beginning and end of the study³⁴. One of the vegetated plots had a canopy cover of 16% and the second vegetated plot had a canopy cover of 32% (Fig. S1). For comparison, canopy cover in undisturbed vegetated areas near the tailings pile is approximately 30%. Neither of the two control plots received a compost amendment. One was irrigated while the second was not irrigated. Neither control plot had any vegetation.

2.3. Dust measurements and metal(loid)s analysis (passive sampling)

Measurements of horizontal dust flux were obtained at the upwind and downwind edges of the four study plots using a series of Modified Wilson & Cooke (MWAC) samplers⁴⁰. The MWAC samplers were installed on a vertical mast with five individual dust collectors on each mast located at 0.06, 0.18, 0.25, 0.50, and 1.0 m above the soil surface (Fig. S3 A). The distance between each set of upwind and downwind sampler masts was approximately 15 m and was oriented south to north based on the predominant wind direction (Fig. S1). The MWAC samplers were constructed using 125-mL wide-mouth high-density polyethylene (HDPE) bottles⁴⁰. The HDPE bottles had an inlet and outlet tube secured to the lid with inside diameter of 7.5 mm and an outer diameter of 10 mm on each sampler (Fig. S3 B). The inlet tube was oriented to the predominant wind direction (Fig. S3 C).

Dust samples were collected during three monitoring periods: May – September 2011; September 2011 – January 2012, and January – May 2012. The duration of each event was 109, 123 and 132 days, respectively, and wind speed and wind direction were measured (Fig. S2). Since the three sampling periods span a full year, the results were employed to determine annual average dust fluxes in and out of the plots studied, as detailed below. Only one sampling was performed for each monitoring period and each plot studied.

Following collection, trapped dust was transferred from each sampler to a glass vial. Dust samples were collected by triple rinsing the MWAC samplers with DI water and collecting the leachate in 30-mL borosilicate glass vials with screw caps. After collection, dust samples were transported to the lab and oven-dried at 60 °C until constant weight was achieved (~ 48 h) and the mass for each height sampler recorded.

Metal(loid) analysis was conducted using a 20-mg aliquot of dust collected from the five MWAC samplers on each mast. Twenty mL of *aqua regia* (1.03 M HNO₃/2.23 M HCl, trace-metal grade) were added to the dust aliquots, which were then placed into a sonicating water bath at 80 °C for 60 min⁴¹. Samples were diluted as needed and analyzed using inductively coupled plasma mass spectrometry (Agilent 7700X with and Octopole Reaction System) for 10 metal(loid)s: aluminum (Al), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), Iron (Fe), magnesium (Mg), manganese (Mn), and lead (Pb) at the University of Arizona Laboratory for Emerging Contaminants. Certified calibration standards from Accustandard were made with MiliQ water, 0.669 M HCl (Fisher, trace-metal grade) and 0.309 M HNO₃ (EMD, Omnitrace). National Institute of Standards and Technology (NIST) standard reference material (SRM 1643e trace elements in water) was analyzed with each set of samples.

2.3.1 Horizontal dust flux calculation—The flux of dust into each individual sampler was calculated as shown in Eq. 1: where mass is in g, the orifice area of the sampler is 0.045 m² and the collection time in days (d) or years (y) depending on the duration of the sampling event.

$$Dust\ Flux = \frac{mass\ dust\ collected}{inlet\ orifice\ area\ (m^2) \times collection\ time\ (y)} \quad (Eq. 1)$$

The horizontal dust flux (g m⁻² y⁻¹) between each discrete height (x = 0.06, 0.18, 0.25, 0.5, or 1.0 m) set of upwind and downwind samplers was calculated by difference (Eq. 2).

$$Horizontal\ Dust\ Flux = Dust\ Flux\ Out_x - Dust\ Flux\ In_x \quad (Eq. 2)$$

The average dust flux (g m⁻² y⁻¹) over the entire height range measured was estimated by fitting the dust flux values calculated using Eq. 1 to an exponential function (Eq. 3)⁴²⁻⁴⁴. Fits of the data gave an average r² of 0.78, ranging from 0.69 to 0.86 (Table S2; Fig. S4). Associated p values showed significant fits at the 0.05 level for all except two of the treatments; the 16% canopy cover incoming flux (p = 0.069) and the 32% canopy cover outgoing flux (p = 0.131).

$$f(x) = ae^{-bx} \quad (Eq. 3)$$

where *a* and *b* are fitting coefficients. The vertical average of the dust flux from 0 to 1 m is

$$Vertical\ Average\ Dust\ Flux = \bar{f} = \frac{a}{bh}(1 - e^{-bh})$$

The average net horizontal dust flux was then calculated using Eq. 4:

$$\text{Average Net Horizontal Dust Flux} = \text{Vertical Average Dust Flux Out} - \text{Vertical Average Dust Flux In} \quad (\text{Eq. 4})$$

where positive values for the net dust flux represent dust production or dust emission from the plot and negative values represent deposition into the plot. The annual average net horizontal dust flux was calculated as the arithmetic average of the fluxes for the three sampling periods (May – Sep; Sept – Jan; Jan – May).

2.4 Short term dust monitoring (DustTrak)

Six DustTrak aerosol monitors (TSI Inc. DRX 8532) were deployed to measure dust concentrations in study areas 1-3. DustTraks were placed at the upwind and downwind edges of the study plots, separated by a distance of approximately 15 m. DustTraks provide simultaneous real-time mass readings (mg m^{-3}) for PM_{10} , $\text{PM}_{2.5}$, PM_{4} , PM_{10} , and TSP ($37 \mu\text{m}$)⁴⁵. Sampling events occurred in 2011 on May 24 following the start of early summer plant growth, and on June 6 during a period of maximum canopy density. Sampling was performed when winds were strongest during the day, typically between 11:00 and 18:00 Mountain Standard Time, and when prevailing winds were from the west-southwest (May 24th) and south-southwest (June 6th). Wind speed was measured within each of the study plots using a series of six Kestrel Weather Meters (Nielsen Kellerman 4500; Table S3; Fig. S5). The DustTrak monitors were housed in an Environmental Enclosure (TSI Inc. 8535) with an omnidirectional inlet resulting in a cut-point diameter of $37 \mu\text{m}$. The flow rate was 3.0 L min^{-1} for the duration of the study period and measurements were taken with five-second resolution. The sample inlet for the DustTrak collectors was set at 1 m above the soil surface within each of the study plots. The DustTrak sampling period was four hours which was subsequently divided into sub-intervals of 15 min. Average measured concentrations and standard deviations for the sixteen 15-min periods were calculated.

3. Results

3.1. Passive dust sampling

In this study, we examined the effectiveness of a plant canopy cover developed during a compost-assisted phytoremediation field trial³⁴ on reducing dust emissions from metal(loid)-contaminated mine tailings. The annual average net horizontal dust flux was determined for four study areas in the field trial including one with 32% canopy cover, one with 16% canopy cover, an irrigated control, and a control that received no irrigation. Dust was collected during three consecutive sampling periods over the course of one year. The annual average net horizontal dust flux was calculated by integrating flux measurements taken at 0.06, 0.18, 0.25, 0.5, and 1.0 m above the tailings surface and then averaging the results from the three sampling periods. Results indicate sparse (16%) vegetative cover resulted in an average dust deposition rate of $371 \text{ g m}^{-2} \text{ y}^{-1}$, whereas the denser (32%) vegetative cover was nearly twice as effective, with an average dust deposition of $606 \text{ g m}^{-2} \text{ y}^{-1}$ (Fig. 2, Table S4, Table S5). These results are in contrast to the control plots, which were characterized by net dust emission rather than dust deposition. Average annual emission

from the irrigated control plot was $1,512 \text{ g m}^{-2} \text{ y}^{-1}$ while emission from the unirrigated control plot exceeded $2,323 \text{ g m}^{-2} \text{ y}^{-1}$.

Horizontal dust flux across the windward edge of the study plots was similar for all treatments (Table S4 and S5). This suggests that the observed differences in dust emission and deposition rates among the treatments are caused primarily by the amount of horizontal dust flux emitted from the leeward edge of study plots as opposed to inherent differences in the concentration of dust blowing into the plots. Horizontal dust flux measurements were greatest close to the surface of the tailings and decreased monotonically with height up to 1.0 m above the surface (Fig. 3; Table S6). Horizontal dust flux emitted from the leeward edge of the study plots was greatest in the control plots (Table S5).

In terms of seasonality, the dense vegetation cover (32%) was most effective at reducing dust emissions during the fall months (Table S5), corresponding to the time of peak biomass in the growing season. Dust deposition in the study area with 16% vegetation cover was greatest during the spring months, indicating that even a small amount of vegetation cover could substantially increase rates of dust deposition under conditions that are typically more favorable for emissions (i.e., during times of low relative humidity with strong prevailing winds)⁸. In contrast, dust emissions were greatest in the irrigated and non-irrigated control plots during the spring months and were nearly twice as high as the emissions rates observed during the summer months when relative humidity is greatest, highlighting the role of atmospheric humidity in reducing dust emissions, especially in the absence of vegetation^{8,46,47}.

The annual average horizontal dust flux at each of the five sampling heights was also examined separately (Fig. 3, Table S6). The study area with 32% canopy cover showed a decrease in net dust flux with increasing height with no contribution from the 0.5 or 1.0 m heights to deposition. Dust deposition in the 16% canopy cover study area was most influenced by the 0.25 and 0.5 m heights with little contribution from the 0.06, 0.18, or 1.0 m heights. For the irrigated control study area, the heights with greatest dust emission were 0.25 and 0.5 m with little contribution from the other heights. Similarly, for the unirrigated control the greatest dust emission was at the 0.25 and 0.5 m heights but in this case there was also a contribution from the 0.06 and 0.18 heights as well.

3.2. Metal(loid)s associated with dust flux

The concentrations of As, Cd, Cu and Pb, in the dust at the IKMHSS site were measured at the upwind and downwind edges of the four study plots (Fig. 4). The annual average net horizontal flux follows a similar pattern for each of the metal(loid)s reported, with the highest concentrations in the unirrigated control, followed by the irrigated control, the 16% canopy cover, and the 32% canopy cover study areas. The 32% canopy cover generally resulted in net deposition of metal(loid)s in the study plot, in particular for As. The deposition of metal(loid)s was not observed in the other treatments, where the net effect of horizontal dust flux on the tailings resulted in the loss and transport of metal(loid)s from the study plots. The net horizontal flux values associated with the metal(loid)s are proportional to the concentrations in the tailings, 2,590, 2,200, 127, and 7.1 mg kg^{-1} , for As, Pb, Cu, and Cd, respectively (Table S7).

3.3. DustTrak particle size-resolved measurements

DustTrak temporal high resolution measurements were conducted in three of the four study areas; 16%, 32% canopy cover, and irrigated control treatment. In general, a net deposition of dust was observed in the three study areas across all particle sizes of dust measured, with the exception of PM₃₇ in the 32% canopy plot (Fig. 5). There were large variances in dust concentrations during the both monitoring periods (particularly for the May monitoring period) that were reflected in large standard deviations in the average measurements. These variations are a consequence of changes of wind speed between the short-time (15 min) measuring periods (see discussion below).

4.0 Discussion

4.1 Factors affecting dust flux

Vegetation established using compost-assisted phytoremediation reduced dust emissions and contaminant transport from mine tailings, as demonstrated by passive dust sampling, which provides a measurement of dust flux over long periods (weeks to months). This helps to average out fluctuations in meteorological parameters such as wind speed and direction, relative humidity, and precipitation^{40,48,49}.

Previous studies have shown that both relative humidity and wind speed are important determinants of dust emissions^{8,12,50} and that local topography and wind velocity profiles are likely the major factors controlling dust deposition at IKMHSS and other similar sites^{19,26,51,52} (Table S1; Table S3; Table S5). To our knowledge, there are no studies that report the effect of vegetation on dust flux for mine tailings, but results can be compared to similar ecosystems. Net horizontal dust flux (emission) rate of approximately 1800 g m⁻² y⁻¹ was reported for bare areas in a natural semiarid ecosystem, located 50 miles south of Tucson, AZ²⁶. In contrast, there was a net horizontal flux (deposition) rate of 900 - 2700 g m⁻² y⁻¹ for herbaceous-dominated and shrub-dominated patches, respectively. These flux values are very similar to those reported here for mine tailings although the canopy cover levels were higher, generally > 60%, in the previous study.

Previous work has shown that canopy distribution, density, and composition are important factors in protecting the surface from the impact of saltating particles^{26,29,53,54}. Control of saltation is critical for controlling dust emissions because wind speed alone is often insufficient to overcome interparticle adhesion forces and release dust from soil surfaces^{25,55-57}. Out of the four study areas examined, results suggest that the 32% canopy cover was the only one that effectively reduced saltation. In this case, there was substantial dust deposition of dust right above the tailings surface (Fig. 3).

Vegetation cover can also act to intercept fine dust particles in suspension and increase rates of dust deposition on the surfaces of leaves^{57,58}. Although this study did not quantitatively address the mass of dust collected on leaves, it was observed that leaves collected dust. Dust layers were less apparent after rainfall events suggesting that accumulated dust was washed onto the tailings surface.

Part of the dust deposition effect observed in this study is likely due to compost application alone, which changes the structure of the topsoil. The amount, stability, and placement of organic matter on a tailings surface is a factor affecting the susceptibility of the surface to wind erosion²⁵. The incorporation of organic matter into the surface of the tailings can lead to an increase in soil aggregate stability⁵⁹, which can substantially reduce the potential for wind erosion.

The tailings at the IKMHSS study site have physical crusts and large patches of efflorescent salts on the surface that contain high levels of toxic metal(loid)s including As, Pb, Cu, and Cd⁵⁶. The dynamic nature and heterogeneous distribution of these physical crusts and efflorescent salts on the surface of the tailings likely contributed to the observed differences in the average concentration of metal(loid)s in dust collected⁵⁶ at the windward and leeward edges of the study areas. In general, the concentration of metal(loid)s in dust (Al, As, Cd, Cr, Cu, Fe, Mg, Mn, and Pb,) was highly dynamic across all treatments both at small spatial scales on the order of several meters and temporally, with no apparent trends or significant differences detected during the one year monitoring period (Table S7). For example, dust samples collected from masts located within a few meters downwind of efflorescent salts typically had higher levels of metal(loid)s than dust samples collected downwind of physical crusts.

4.2 Metal(loid) transport

Results indicate that vegetation had no effect on the average concentration of metal(loid)s in the dust in either the 16% or 32% canopy cover plots. But vegetation did have an effect on the mass fluxes of contaminants due to the reduction in net dust flux (Fig. 4, Table S7). Previous studies using Micro-Orifice Uniform Deposit Impactor (MOUDI) and a laboratory dust generation and fractionation system showed that metal(loids) in windblown dust from the tailings are present at higher concentrations in the sub-micron particle size range^{5,9}. The results suggest that MWAC samples at the plot exits are not enriched in the sub-micron particle size range. In sulfide mineral deposits, such as the IKMSS, crystals with a high concentration of metal(loids) will form after rain events and evapotranspiration process⁵. These metal(loid)s enriched crystals are more susceptible to break in smaller particles and subsequent be transported by wind².

4.3 Short-term measurements

An array of DustTrak monitors was used to collect simultaneous high temporal resolution measurements of dust flux across a range of particles sizes to attempt to quantify potential reduction in dust emissions during strong wind events. However, short-term measurements are affected by large variability in meteorological conditions such as wind speed and direction (Table S3; Fig. S5) so that dust emission and deposition may change rapidly during the sampling period. For this reason, some of the DustTrak results did not follow the expected trend. For example, the net deposition of PM₃₇ was greatest in the irrigated control plot during the May wind event, whereas the plot with the dense vegetation cover had the highest rates of dust emission, which was observed during the June wind event. The variability imposed by short-term changes in meteorological conditions is reflected in the large standard deviations of dust concentration differences (error bars in Fig. 5).

In summary, this study quantified the reduction in dust emissions from a mine tailings site due to vegetation established using compost-assisted phytoremediation. The direct benefit of the vegetative cap was a reduction in windborne tailings particulates and associated metal(loid)s entering the surrounding environment. A second benefit is an attendant reduction in human health risks associated with inhalation or ingestion exposures following the deposition of dust particles onto surfaces, food, and soils in the neighboring community of Dewey-Humboldt¹⁶. Quantitative information such as provided by this study is needed for incorporation into risk assessment models and to help evaluate phytoremediation as a reclamation strategy for mine tailings.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

ACKNOWLEDGMENTS

This research was supported by NIEHS Superfund Research Program Grant P42 ES004940. We thank Steven Schuchardt, president of North American Industries, for providing access to the IKMHSS site and on-going help with irrigation and the weather station. We thank Dr. David Breshears for providing access to field equipment, Mary K. Amistadi at ALEC for help with metal(loid) analysis as well as, Kyle P. Rine and MacKenzie R. Russell from The Department of Hydrology and Atmospheric Sciences, for field work support. The views of authors do not necessarily represent those of the NIEHS, NIH.

REFERENCES

- (1). Corriveau MC; Jamieson HE; Parsons MB; Campbell JL; Lanzarotti A Direct characterization of airborne particles associated with arsenic-rich mine tailings: Particle size, mineralogy and texture. *Appl. Geochemistry* 2011, 26 (9–10), 1639–1648.
- (2). Meza-Figueroa D; Maier RM; de la O-Villanueva M; Gómez-Alvarez A; Moreno-Zazueta A; Rivera J; Campillo A; Grandlic CJ; Anaya R; Palafox-Reyes J The impact of unconfined mine tailings in residential areas from a mining town in a semi-arid environment: Nacozari, Sonora, Mexico. *Chemosphere* 2009, 77 (1), 140–147. [PubMed: 19500816]
- (3). Ye ZH; Shu WS; Zhang ZQ; Lan CY; Wong MH Evaluation of major constraints to revegetation of lead/zinc mine tailings using bioassay techniques. *Chemosphere* 2002, 47 (10), 1103–1111. [PubMed: 12137044]
- (4). Root RA; Hayes SM; Hammond CM; Maier RM; Chorover J Toxic metal(loid) speciation during weathering of iron sulfide mine tailings under semi-arid climate. *Appl. Geochemistry* 2015, 62, 131–149.
- (5). Hayes SM; Root RA; Perdrial N; Maier RM; Chorover J Surficial weathering of iron sulfide mine tailings under semi-arid climate. *Geochim. Cosmochim. Acta* 2014, 141, 240–257. [PubMed: 25197102]
- (6). Hudson-Edwards KA; Jamieson HE; Lottermoser BG Mine Wastes: Past, Present, Future. 2011, 7 (6), 375–380.
- (7). Lottermoser BG Mine wastes characterization, treatment and environmental impacts; Springer: Berlin; London, 2010.
- (8). Csavina J; Field J; Félix O; Corral-Avitia AY; Sáez AE; Betterton EA Effect of wind speed and relative humidity on atmospheric dust concentrations in semi-arid climates. *Sci. Total Environ* 2014, 487 (1), 82–90. [PubMed: 24769193]
- (9). Csavina J; Landázuri A; Wonaschütz A; Rine K; Rheinheimer P; Barbaris B; Conant W; Sáez AE; Betterton EA Metal and metalloid contaminants in atmospheric aerosols from mining operations. *Water. Air. Soil Pollut* 2011, 221 (1–4), 145–157. [PubMed: 23441050]

- (10). Mendez MO; Maier RM Phytostabilization of mine tailings in arid and semiarid environments - An emerging remediation technology. *Environ. Health Perspect* 2008, 116 (3), 278–283. [PubMed: 18335091]
- (11). Reis AP; da Silva EF; Fonseca EC; Patinha C; Barrosinho C; Matos J Environmental Assessment of the Caveira Abandoned Mine (Southern Portugal): Part 1: Characterization of Metal Contaminated Soil. *Soil Sediment Contam.* 2012, 21 (2), 227–254.
- (12). Stovern M; Guzmán H; Rine KP; Felix O; King M; Ela WP; Betterton EA; Sáez AE Windblown dust deposition forecasting and spread of contamination around mine tailings. *Atmosphere (Basel)*. 2016, 7 (2), 16. [PubMed: 29082035]
- (13). Jianu D; Iordache V; Soare B; Petrescu L; Neagoe A; Iacob C; Orza R The role of mineralogy and geochemistry in hazard potential assessment of mining areas In *Bio-Geo interaction in contaminated soils*; Kothe E, Varma A, Eds.; Springer Berlin Heidelberg, 2012; pp 35–79.
- (14). Kim J; Kim K-W; Ahn J-S; Ko I; Lee C-H Investigation and risk assessment modeling of As and other heavy metals contamination around five abandoned metal mines in Korea . *Environ. Geochem. Health* 2005, 27 (2), 193–203. [PubMed: 16003587]
- (15). Ramirez-Andreotta MD; Brusseau ML; Artiola JF; Maier RM A greenhouse and field-based study to determine the accumulation of arsenic in common homegrown vegetables grown in mining-affected soils. *Sci. Total Environ* 2013, 443, 299–306. [PubMed: 23201696]
- (16). Csavina J; Field J; Taylor MP; Gao S; Landázuri A; Betterton EA; Sáez AE A review on the importance of metals and metalloids in atmospheric dust and aerosol from mining operations. *Sci. Total Environ* 2012, 433, 58–73. [PubMed: 22766428]
- (17). Stovern M; Rine KP; Russell MR; Félix O; King M; Eduardo Sáez A; Betterton EA Development of a dust deposition forecasting model for mine tailings impoundments using in situ observations and particle transport simulations. *Aeolian Res.* 2015, 18, 155–167.
- (18). Loh MM; Sugeng A; Lothrop N; Klimecki W; Cox M; Wilkinson ST; Lu Z; Beamer PI Multimedia exposures to arsenic and lead for children near an inactive mine tailings and smelter site. *Environ. Res.* 2016, 146, 331–339. [PubMed: 26803211]
- (19). Brotons JM; Díaz AR; Sarriá FA; Serrato FB Wind erosion on mining waste in southeast Spain. *L. Degrad. Dev* 2010, 21 (2), 196–209.
- (20). Taylor MP; Mackay AK; Hudson-Edwards KA; Holz E Soil Cd, Cu, Pb and Zn contaminants around Mount Isa city, Queensland, Australia: Potential sources and risks to human health. *Appl. Geochemistry* 2010, 25 (6), 841–855.
- (21). McKenna Neuman C; Boulton JW; Sanderson S Wind tunnel simulation of environmental controls on fugitive dust emissions from mine tailings. *Atmos. Environ.* 2009, 43 (3), 520–529.
- (22). Griffin DW; Kellogg CA; Shinn EA Dust in the Wind: Long Range Transport of Dust in the Atmosphere and Its Implications for Global Public and Ecosystem Health. *Glob. Chang. Hum. Heal* 2001, 2 (1), 20–33.
- (23). Kok JF; Parteli EJR; Michaels TI; Karam DB The physics of wind-blown sand and dust. *Reports Prog. Phys* 2012, 75 (10), 106901.
- (24). Stout JE; Zobeck TM The Wolfforth field experiment: A wind erosion study. *Soil Sci.* 1996, 161 (9), 616–632.
- (25). Ravi S; D 'odorico P; Breshears DD; Field JP; Goudie AS; Huxman TE; Li J; Okin GS; Swap RJ; Thomas AD; et al. Aeolian Processes and the Biosphere. *Rev. Geophys* 2011, No. 2010, 1–45.
- (26). Field JP; Breshears DD; Whicker JJ; Zou CB Sediment capture by vegetation patches: Implications for desertification and increased resource redistribution. *J. Geophys. Res. Biogeosciences* 2012, 117 (1), 1–9.
- (27). Zobeck TM; Sterk G; Funk R; Rajot JL; Stout JE; Pelt RS Van. Measurement and data analysis methods for field-scale wind erosion studies and model validation. *Earth Surf. Process. Landforms* 2003, 28 (11), 1163–1188.
- (28). Field JP; Belnap J; Breshears DD; Neff JC; Okin GS; Whicker JJ; Painter TH; Ravi S; Reheis MC; Reynolds RL The ecology of dust. *Front. Ecol. Environ* 2010, 8 (8), 423–430.

- (29). Breshears DD; Whicker JJ; Zou CB; Field JP; Allen CD A conceptual framework for dryland aeolian sediment transport along the grassland-forest continuum: Effects of woody plant canopy cover and disturbance. *Geomorphology* 2009, 105 (1–2), 28–38.
- (30). Burri K; Gromke C; Lehning M; Graf F Aeolian sediment transport over vegetation canopies: A wind tunnel study with live plants. *Aeolian Res.* 2011, 3 (2), 205–213.
- (31). Nordstrom K; Hotta S Wind erosion from cropland solutions in the USA: a review of problems, and prospects. *Geoderma* 2004, 121 (3–4), 157–167.
- (32). Ulriksen C; Ginocchio R; Mench M; Neaman A Lime and Compost promote plant re-colonization of metal-polluted, acidic soils. *Int. J. Phytoremediation* 2012, 14 (8), 820–833. [PubMed: 22908647]
- (33). Santibañez C; De La Fuente LM; Bustamante E; Silva S; León-Lobos P; Ginocchio R Potential use of organic- and hard-rock mine wastes on aided phytostabilization of large-scale mine tailings under semiarid mediterranean climatic conditions: Short-Term field study. *Appl. Environ. Soil Sci* 2012, 2012, 1–15.
- (34). Gil-Loaiza J; White SA; Root RA; Solís-Dominguez FA; Hammond CM; Chorover J; Maier RM Phytostabilization of mine tailings using compost-assisted direct planting: Translating greenhouse results to the field. *Sci. Total Environ* 2016, 565, 451–461. [PubMed: 27183459]
- (35). USEPA. Remedial Investigation Report Iron King Mine - Humboldt Smelter Superfund Site Dewey-Humboldt, Yavapai County, Arizona United States Environmental Protection Agency; 2010.
- (36). Solís-Dominguez FA; White SA; Hutter TB; Amistadi MK; Root RA; Chorover J; Maier RM Response of key soil parameters during compost-assisted phytostabilization in extremely acidic tailings: Effect of plant species. *Environ. Sci. Technol* 2012, 46, 1019–1027. [PubMed: 22191663]
- (37). Vankat JL Interior Chaparral Shrubland, *in* Vegetation Dynamics on the Mountains and Plateaus of the American Southwest, Vol. 8; Springer, Dordrecht, 2013.
- (38). USEPA. Biological Evaluation of the Iron King Mine-Humboldt Smelter Superfund Site, EnviroSystems Management Project and Report No. 1432–08. United States Environmental Protection Agency; USA, 2009.
- (39). Stovern M; Felix O; Csavina J; Rine KP; Russell MR; Jones RM; King M; Betterton EA; Sáez AE Simulation of windblown dust transport from a mine tailings impoundment using a computational fluid dynamics model. *Aeolian Res.* 2014, 14, 75–83. [PubMed: 25621085]
- (40). Goossens D; Offer ZY Wind tunnel and field calibration of six aeolian dust samplers. *Atmos. Environ* 2000, 34 (7), 1043–1057.
- (41). Harper SL; Walling JF; Holland DM; Pranger LJ Simplex Optimization of Multielement Ultrasonic Extraction of Atmospheric Particulates. *Anal. Chem* 1983, 55 (9), 1553–1557.
- (42). Gillette DA; Fryrear DW; Gill TE; Ley T; Cahill TA; Gearhart EA Relation of vertical flux of particles smaller than 10 μm to total aeolian horizontal mass flux at Owens Lake. *J. Geophys. Res* 1997, 102 (D22), 26009–26015.
- (43). Ellis JT; Li B; Farrell EJ; Sherman DJ Protocols for characterizing aeolian mass-flux profiles. *Aeolian Res.* 2009, 1 (1–2), 19–26.
- (44). Stout JE Dust and environment in the Southern High Plains of North America. *J. Arid Environ* 2001, 47 (4), 425–441.
- (45). Wang X; Chancellor G; Evenstad J; Farnsworth JE; Hase A; Olson GM; Sreenath A; Agarwal JK A Novel Optical Instrument for Estimating Size Segregated Aerosol Mass Concentration in Real Time. *Aerosol Sci. Technol* 2009, 43 (9), 939–950.
- (46). Neuman C; Sanderson S Humidity control of particle emissions in aeolian systems. *J. Geophys. Res. Earth Surf* 2008, 113 (2), 1–10.
- (47). Ravi S; D’Odorico P A field-scale analysis of the dependence of wind erosion threshold velocity on air humidity. *Geophys. Res. Lett* 2005, 32 (21), 1–4.
- (48). Goossens D Relationships between horizontal transport flux and vertical deposition flux during dry deposition of atmospheric dust particles. *J. Geophys. Res. Earth Surf* 2008, 113 (2), 1–13.

- (49). Mendez MJ; Funk R; Buschiazzi DE Field wind erosion measurements with Big Spring Number Eight (BSNE) and Modified Wilson and Cook (MWAC) samplers. *Geomorphology* 2011, 129 (1–2), 43–48.
- (50). Ravi S; D’Odorico P; Over TM; Zobeck TM On the effect of air humidity on soil susceptibility to wind erosion: The case of air-dry soils. *Geophys. Res. Lett* 2004, 31 (9), 2–5.
- (51). Munson SM; Belnap J; Okin GS Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. *Proc. Natl. Acad. Sci. U. S. A* 2011, 108 (10), 3854–3859. [PubMed: 21368143]
- (52). Li J; Okin GS; Alvarez L; Epstein H Quantitative effects of vegetation cover on wind erosion and soil nutrient loss in a desert grassland of southern New Mexico, USA. *Biogeochemistry* 2007, 85 (3), 317–332.
- (53). Wolfe SA; Nickling WG The protective role of sparse vegetation in wind erosion. *Prog. Phys. Geogr* 1993, 17 (1), 50–68.
- (54). Webb NP; Herrick JE; Duniway MC Ecological site-based assessments of wind and water erosion: Informing accelerated soil erosion management in rangelands. *Ecol. Appl* 2014, 24 (6), 1405–1420. [PubMed: 29160663]
- (55). Okin GS; Gillette DA; Herrick JE Multi-scale controls on and consequences of aeolian processes in landscape change in arid and semi-arid environments. *J. Arid Environ.* 2006, 65 (2), 253–275.
- (56). Gonzales P; Felix O; Alexander C; Lutz E; Ela W; Eduardo Sáez A Laboratory dust generation and size-dependent characterization of metal and metalloid-contaminated mine tailings deposits. *J. Hazard. Mater* 2014, 280, 619–626. [PubMed: 25222928]
- (57). Zhu D; Gillies JA; Etyemezian V; Nikolich G; Shaw WJ Evaluation of the surface roughness effect on suspended particle deposition near unpaved roads. *Atmos. Environ* 2015, 122, 541–551.
- (58). Chaturvedi RK; Prasad S; Rana S; Obaidullah SM; Pandey V; Singh H Effect of dust load on the leaf attributes of the tree species growing along the roadside. *Environ. Monit. Assess* 2013, 185 (1), 383–391. [PubMed: 22367367]
- (59). Zanuzzi A; Arocena JM; van Mourik JM; Faz Cano A Amendments with organic and industrial wastes stimulate soil formation in mine tailings as revealed by micromorphology. *Geoderma* 2009, 154 (1–2), 69–75.

Figure 1. Illustration of various mining activities, particle sizes generated, the temporal scale at which particles could be generated or transported and the two dust monitoring approaches (DustTrak and passive sampling) used in this study.

Figure 2.

Effect of treatment on the annual average of the differential horizontal dust flux. Negative numbers indicate there was a net deposition of dust into the study area while positive numbers indicate that there was a net emission of dust from the study area. Windrose insert describes predominant wind direction from May 2011 to May 2012 (WS= wind speed).

Figure 3. Effect of treatment on annual average of the differential horizontal dust flux at 0.06, 0.18, 0.25, and 0.5 and 1.0 m above the mine tailings surface. Negative numbers indicate there was a net deposition (Dep) of dust into the study area while positive numbers indicate that there was a net emission (Em) of dust from the study area.

Figure 4. Effect of treatment on the annual average of the horizontal elemental flux associated with dust particles. Negative numbers indicate there was a net deposition of dust into the study area while positive numbers indicate that there was a net emission of dust from the study area. Metals analyzed were arsenic (As), lead (Pb), copper (Cu) and cadmium (Cd).

Figure 5.

Effect of treatment on dust flux for different particle sizes. DustTrak data were analyzed by averaging data from 15 min intervals of a four hour measurement time. Values presented are the average $\pm$ standard deviation ($n=16$); negative numbers indicate there was a net deposition of particle matter into the study area while positive numbers indicate that there was a net emission of particle matter. Windrose inserts show predominant wind direction (WS= wind speed). Rates of dust emission and/or deposition were estimated for each of the study plots by DustTrak Flux out – Flux in. Wind roses measuring day average wind direction (vectors) and speed (m s^{-1}) for May 24th, 2011 and June 6th, 2011. Differences show the variability of weather conditions in different seasons.