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Development of vegetation based soil quality indices for mineralized terrane in arid and semi-arid regions

S.W. Blecker[a,*], L.L. Stillings[a], M.C. Amacher[b], J.A. Ippolito[c], N.M. DeCrappeo[d]

[a] USGS, 1664 N. Virginia St., MS-176, Reno, NV 89557, United States
[b] US-Forest Service, 860N 1200E, Logan, UT 84321, United States
[c] USDA-ARS, 3793N 3600E, Kimberly, ID 83341, United States
[d] USGS, 3200 SW Jefferson Way, Corvallis, OR 97331, United States

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ABSTRACT

Soil quality indices (SQIs) are often management driven and attempt to describe key relationships between above- and below-ground parameters. In terrestrial systems, indices that were initially developed and modified for agroecosystems have been applied to non-agricultural systems in increasing number. We develop an SQI in arid and semi-arid ecosystems of the Western US impacted by different types of geologic mineralization using the relationship between vegetation community parameters and soil abiotic and biotic properties. We analyze these relations in soils associated with three different mineralization types: podiform chromite, Cu/Mo porphyry, and acid-sulfate gold vein systems at four different sites in California and Nevada. Soil samples were collected from undisturbed soils in both mineralized and nearby unmineralized substrates as well as from waste rock and tailings. Aboveground net primary productivity (ANPP), canopy cover and shrub density were measured for the vegetative communities. Minimum data sets were developed based on correlations between the soil and vegetation parameters, refined using principal components analysis, scored using non-linear functions, and combined into an overall SQI. The indices are comprised of one or two microbial parameters and three to six abiotic parameters, the latter consisting of nutrients and metals. Given the preliminary development of this approach, the parameters and combinations to arrive at an SQI for a given site cannot at this time be correlated or compared with that of another site. This SQI approach provides a means of quantifying disturbed ecosystem recovery resulting from mining, and could be applied to other disturbances in a way that readily distills the information for potential use by land managers. However, severely disturbed areas with little to no aboveground biomass, such as unreclaimed tailings, have likely crossed an ecological threshold that precludes the use of this type of monitoring tool.

1. Introduction

The myriad definitions of soil or ecosystem quality are often management driven and typically focus on the ability of soil to provide functions relating to biological productivity and environmental quality (Doran and Parkin, 1994; Karlen et al., 1997). A variety of attempts have been made to quantify the complexities of soil quality and provide a means of evaluating the impact of natural and anthropogenic disturbances. Though not without their limitations (Sojka and Upchurch, 1999), such indices can improve our understanding of the controls behind ecosystem processes and help bridge the gap between scientific and regulatory communities.

In terrestrial systems, indices were initially developed and modified for agroecosystems (Doran and Parkin, 1994), but the number of studies implementing such indices in non-agricultural systems is growing (Bastida et al., 2008). Regardless of application scenario, soil quality indices (SQIs) are generally comprised of a mixture of biological, physical, and chemical parameters that attempt to distill the complexity of a system, through various correlative processes, into a metric of a soil's or ecosystem's ability to carry out one or more functions (Papendick and Parr, 1992; Halvorson et al., 1996). Studies utilizing simple ratios, based on relationships of parameters such as the metabolic quotient, quantity of mineralized substrate/unit of microbial biomass carbon/unit of time (qCO_2) or enzyme activity/total C, are generally too simplistic for interpretations of soil quality (Gil-Sotres et al., 2005). However these same measurements, when combined with others, tend to form the core of many SQIs.

Effective SQIs should correlate well with soil or ecosystem processes such as nutrient cycling and vegetative productivity, integrate those properties and processes, and be responsive to management practices (Doran and Parkin, 1996; Dalal, 1998; Nortcliff,

* Corresponding author. Tel.: +1 916 445 0464; fax: +1 775 784 5079.
E-mail address: sblecker@deltacouncil.ca.gov (S.W. Blecker).

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2002). Cost, accessibility, ease of interpretation, and availability of existing data often dictate parameter selection. Commonly applied biological parameters are associated with microbial activity or function (e.g., mineralization, respiration, microbial biomass, enzyme activity; Winding et al., 2005). Abiotic parameters (e.g., soil texture, nutrient status, total carbon as affected by site management) can further provide context to more clearly interpret the biological measures.

A multi-parametric index for agroecosystems developed by Karlen et al. (1997) employed a framework of weighted and integrated soil parameters related to management goals such as plant productivity and maintenance of environmental quality. This concept has been developed utilizing selections of various soil quality indicators, integrated into an index of soil quality for agroecosystems (Andrews and Carroll, 2001; Sharma et al., 2005; Masto et al., 2008), forests (Bastida et al., 2008), and rangelands (Rezaei et al., 2006). However, SQIs for ecosystems disturbed by mining are less common (Burger et al., 2010).

Surface mining often causes severe disturbance, with drastic alteration of physical, chemical, and biological properties (McSweeney and Jansen, 1984; Insam and Domsch, 1988; Shukla et al., 2004). Misleading evidence of recovery obtained from measurements of single properties can often be overcome by measuring a variety of properties (Insam and Domsch, 1988). To that end, numerous researchers have employed principal components analysis to identify key relationships among biological, chemical and physical parameters in reclamation studies (e.g., Bentham et al., 1992; Shukla et al., 2004; Pereira et al., 2006; Vasquez-Murrieta et al., 2006). Mummey et al. (2002) stated that criteria for judging surface mine reclamation success predominantly relies on aboveground indicators, but a more complete picture of ecosystem health should include the activity and composition of soil microbial communities. Ruiz-Jaen and Aide (2005) and Haney et al. (2007) have also stressed the need to include multiple measures of ecosystem attributes such as vegetation diversity and structure, along with belowground processes, that can be compared to reference sites. Using enzyme assays and microbial biomass, Claassens et al. (2008) pointed out the benefit of the additional information provided by measurements of microbial parameters that are not traditionally included with the aboveground data collection associated with reclamation monitoring. The purpose of our study was to apply the SQI concept to identify soil quality indicators in mineralized terrane that could be used to monitor reclamation recovery in areas impacted by surface mining.

2. Methods

2.1. Site descriptions

The study sites covered a range of semi-arid/mineralized ecosystems designed to evaluate the utility and variability of soil quality indicators. For the purposes of this study, a mineralized ecosystem is one where the underlying bedrock has been mineralogically altered by exposure to high-temperature geological fluids. Within a given study site, samples were taken from three areas: (1) unmineralized/undisturbed (no alteration by high-temperature fluids and undisturbed by mining), (2) mineralized/undisturbed (containing metal-bearing minerals, usually sulfides or oxides formed by primary geologic processes or geological alteration associated with or following the mineralization processes, and undisturbed by mining), and (3) mineralized/disturbed (disturbed by previous mining activity). The mineralized/disturbed sites are mine dumps consisting of material which had been removed and transported during mining operations and ranged from waste rock to tailing piles. Aside from mineralization, the material in the mine dumps has also been subjected to numerous physical and chemical alterations such as change in soil structure and density, loss of organic matter and nutrients, and change in pH, which in concert can adversely impact above and belowground biota. As we did not know the exact treatment of the mine dump sediments, they were qualitatively classified as either waste rock or tailings based on field observations and vegetation and soils data. Compared to the tailings, waste rock typically had greater vegetative cover, microbial activity and nutrient concentrations.

2.1.1. Chinese Camp mining district: podiform chromite mineralization

This study site is located south of Jamestown, CA (37.85°N, 120.4°W) at an average elevation of 530 m, a mean annual precipitation (MAP) of 820 mm, and a mean annual air temperature (MAT) of 14.9 °C. The mineralized/undisturbed sample site consists of serpentine soils which developed on ultramafic rocks consisting of dunite (olivine) and peridotites (various mixtures of olivine and clinopyroxene, along with chromite) where Mg-rich silicates have been hydrothermally altered to serpentinite, a hydrous Mg-, Fe-silicate. These soils are typically characterized by low Ca/Mg, relatively high Ni and Cr contents, and often support unique vegetative communities (Kruckeberg, 1984; Proctor, 1999). The combination of unusual soils, and rare and unique flora and fauna, led the Bureau of Land Management (BLM) to designate this area as an Area of Critical Environmental Concern (ACEC) (http://www.blm.gov/ca/st/en/fo/folsom/redhillshomepg1.html, last accessed March, 2011). The unmineralized/undisturbed sample area consisted of an open woodland (*Quercus douglasii*, *Pinus sabiniana*)/annual grass (*Bromus* sp., *Erodium* sp., *Avena fatua*, *Amsinckia menziesii*) community that developed on andesite soils, and provides stark contrast to the adjacent buckbrush chaparral (*Ceanothus cuneatus*) of the serpentine soils. The mineralized/disturbed sample areas consists of waste rock and tailings piles in areas of defunct chromite mines located within the serpentinite system (Blecker et al., 2010).

2.1.2. Battle Mountain mining district: Cu/Mo porphyry mineralization

The Battle Mountain study site is located southwest of Battle Mountain, NV (40.57°N, 117.1°W) at an average elevation of 1380 m, an MAP of 210 mm, and an MAT of 9.5 °C. The mineralization at this site consists of stockwork veinlets of quartz, chalcopyrite, and molybdenite, surrounding a felsic porphyritic intrusion consisting of quartz, biotite, and primary feldspars altered to assemblages of secondary feldspars, chlorite, sericite and various clays, depending on the zonation of the alteration (Theodore et al., 1992). Upon weathering, the sulfides associated with these deposits generate considerable acidic drainage (Nash, 2005). The surrounding non-mineralized rocks are predominantly interbedded sandstones, shales and greenstones. (Theodore et al., 1992) The area was mined throughout the 19th and 20th centuries for Cu, Mo, Au and other metals from chalcopyrite and molybdenite. As the sagebrush communities (*Artemisia* sp.) on both the mineralized and unmineralized rock do not differ visually, we utilized detailed mapping of Theodore et al. (1992) to determine appropriate sampling areas. Waste rock and tailings were sampled in areas of abandoned mines (Blecker et al., 2010).

2.1.3. Castle Peak mining district and Masonic mining district: acid sulfate mineralization

The Castle Peak study site is located east of Reno, NV (39.48°N, 119.7°W), at an average elevation of 1350 m, an MAP of 185 mm, and an MAT of 10.4 °C. The Masonic study site is located northeast of Bridgeport, CA (38.40°N, 119.1°W) at an average elevation of 2125 m, an MAP of 210 mm, and an MAT of 5.0 °C. In these locations, epithermal alteration has resulted in acid-sulfate gold mineralization with quartz, pyrite, alunite and various clay minerals comprising the dominant mineralogy (Castor et al., 2005). The open woodland vegetation (*Pinus jeffreyi, Pinus ponderosa, Eriogonum robustum*) of the mineralized areas provides stark contrast to the sagebrush (*Artemisia* sp.) shrubland of the unmineralized areas that have developed on andesite. Mineralization and subsequent weathering of minerals such as pyrite has resulted in soils of low pH and fertility (Schlesinger et al., 1989). Waste rock and tailings were sampled at abandoned Hg and ferricrete mines at the Castle Peak site, and abandoned gold mines at the Masonic site (Blecker et al., 2010).

2.2. Study design/field sampling

At each of the four study sites, three random plots were selected within each of the three sample areas (unmineralized/undisturbed, mineralized/undisturbed, mineralized/disturbed waste rock or tailings). Each plot within a given sample area was selected to have a similar aspect (150–210°), elevation, and slope. In each of the plots, three 30-m transects (spaced 120° apart) were randomly established. Three soil samples (0–15 cm in depth) were taken at random distances along each transect for a total of 9 samples per area for bulk chemistry and microbial analyses. Separate soil core samples (0–15 cm) were taken immediately adjacent to the other soil samples using a slide-hammer for measurements of bulk density and soil moisture.

The same 30-m transects were used for vegetation measurements. A line-point intercept survey with 0.6-m intervals was conducted along each transect to determine percent canopy cover and percent bare ground ($n = 150$ per plot). In addition, a 4-m belt transect was used to determine densities of trees (categorized by species) and shrubs (categorized as *Ceanothus cuneatus* within the Chinese Camp study site, and *Artemsia* sp., *Chysothamnus* sp., or other, at the other study sites). Aboveground net primary productivity (ANPP) was estimated by harvesting all aboveground living plant material within a 0.5-m quadrat at three random locations along each transect ($n = 9$ per plot). The plant material was oven-dried at 50 °C for 72-h and weighed to obtain estimates of ANPP. Sites were sampled one time in the spring of 2008 near peak soil moisture/microbial activity.

2.3. Soil microbiological, chemical and physical analyses

Parameters related to soil quality were selected to provide a range of indicator variables of both microbial function and structure (Table 1a). Carbon and N nitrogen mineralization potential was performed to provide a general assessment of microbial activity and the ability of the microbial community to generate plant-available N (Robertson et al., 1999). Enzyme assays were performed to analyze the activity of specific soil nutrients and more broad-based enzyme activity. We conducted assays for the S-cycle (arylsulfatase) and P-cycle (acid and alkaline phosphatase) to focus on specific nutrients and fluorescein diacetate (FDA) to measure the activity of a number of different enzymes (protease, lipase, and esterase; Green et al., 2006). All enzyme assays in this study were measurements of potential activity carried out in short-term incubations under controlled temperature and pH and were analyzed spectrophotometrically against a standard curve of known substrate concentrations.

We qualitatively assessed functional diversity of microbial communities with community level physiological profiles (CLPP) using Biolog EcoPlates™ (Biolog Inc., Hayward, CA, USA). Each 96-well EcoPlate contains 31 different C substrates and one water control replicated three times; each well also contains tetrazolium violet dye that, when reduced by microbial utilization of the C substrate, turns the well purple. Average well color development (AWCD) measured at 24-h intervals for 5 consecutive days indicates the microbial community's ability to use a particular substrate, the assumption being that more functionally diverse communities will be able to use more substrates (Sinsabaugh et al., 1999). Day 4 (96-h) readings were used to allow for maximum color development response variance without exceeding the linear absorbance range (Garland, 1996).

Microbial biomass and community structure were measured using a hybrid phospholipid fatty acid (PLFA) and fatty-acid methyl ester (FAME) technique (Smithwick et al., 2005) based on a modified Bligh and Dyer (1959) method. Certain lipid "signatures" within the cell membranes of living microbes can be used to identify a portion of the microbial community: gram+ and gram− bacteria, fungi, actinomycetes, and protozoa (Sinsabaugh et al., 1999). Total microbial biomass-C was estimated by summing abundances of all fatty acids.

Table 1b provides an overview of the methods used to determine abiotic measurements of associated soil chemical (pH, electrical

Table 1a
Measurements of soil microbial activity, biomass, and community structure evaluated for use in a soil quality index (SQI) for mineralized terrane.

Variable	Method
C, N mineralization potential[a,c]	10-d static incubation (samples brought to field capacity with de-ionized water where necessary) with 1 M NaOH trap; CO_2 determined by titration with 1 M HCl; inorganic N determined by 2 M KCl extraction and flow injection analysis (Robertson et al., 1999)
Enzyme activity: S-cycle[b]	Arylsulfatase hydrolysis (Dick et al., 1996); 1-h incubation at 37 °C.
Enzyme activity: P-cycle[b]	Acid (pH = 6.5) and alkaline (pH = 11.0) phosphatase hydrolysis (Dick et al., 1996); 1-h incubation at 37 °C.
Enzyme activity: general[a]	Fluorescein diacetate (FDA) hydrolysis (Green et al., 2006); 1.5-h incubation at 35 °C.
Soil microbial functional diversity[a]	Community-level physiological profiling (CLPP) using Biolog EcoPlates™ (Sinsabaugh et al., 1999)
Soil microbial biomass C and community structure[d]	Phospholipid fatty-acid analysis (PLFA) (Smithwick et al., 2005)

[a] Samples were stored field moist at 4 °C, passed through a 2-mm sieve, and analyzed within 2 weeks of the collection date.
[b] Samples were air-dried, then passed through a 2-mm sieve.
[c] Samples were brought to 60% water-filled pore space just prior to analysis.
[d] Samples were stored field moist at 4 °C; immediately placed in −20 °C storage upon return from the field, freeze-dried within 2 weeks, and analyzed within 3 months.

Table 1b
Metals and relevant physical and chemical parameters evaluated for use in a soil quality index (SQI) for mineralized terrane.

Variable	Method
Particle size distribution[a]	Hydrometer (Elliott et al., 1999)
Bulk density	Soil core (Elliott et al., 1999)
Volumetric moisture content	Gravimetric (oven-dry for 48 h at 110 °C) with bulk density correction
pH[b]	1:2 (soil:de-ionized water) (Thomas, 1996)
Electrical conductivity[a]	Saturated paste extract (Rhoades, 1996)
Total C/N/S[a,c]	Dry combustion; LECO RC-412C species analyzer and LECO TruSpec C/N/S
Metal content[a]	DTPA extractable metals[d] (Amacher, 1996), Total metals (Briggs, 2002; Briggs and Meier, 2002)

[a] Samples were air-dried, then passed through a 2-mm sieve.
[b] Samples were stored field moist at 4 °C, then passed through a 2-mm sieve.
[c] Total organic carbon was determined by subtracting total soil carbon from inorganic carbon.
[d] DTPA (diethylenetriaminepentaacetic acid) extractable metals are generally associated with the bioavailable metals fraction (Amacher, 1996).

Table 2
Means (±1 SEM) of vegetation measurements for all study sites. Similar lower case letters within a column for a given study site are statistically similar at $p < 0.05$.

Study site	Sample area	ANPP[A] ($g\,m^{-2}$)	Canopy cover (%)	Shrub density (plants ha^{-1})
Chinese Camp	Unmineralized/undisturbed (andesite)	28.0[a] (2.8)	71.8[a] (2.7)	n/a[B]
	Mineralized/undisturbed (serpentine)	20.7[a] (2.9)	26.1[b] (2.3)	1990[a] (350)
	Tailings	4.4[b] (0.8)	11.4[b] (0.8)	0[c] (0.0)
	Waste rock	12.2[ab] (1.9)	18.8[b] (0.8)	470[b] (260)
Battle Mountain	Unmineralized/undisturbed (sandstone/arenite)	26.6[a] (5.0)	49.8[a] (7.2)	9990[a] (780)
	Mineralized/undisturbed (Mo/Cu porphyry)	22.4[a] (4.3)	47.7[a] (9.6)	5650[b] (260)
	Tailings	1.2[b] (0.8)	6.5[b] (6.5)	190[d] (66)
	Waste rock	2.5[b] (2.5)	32.0[ab] (14.3)	2900[d] (480)
Castle Peak	Unmineralized/undisturbed (andesite)	22.6[a] (1.0)	54.2[a] (4.1)	8690[a] (1000)
	Mineralized/undisturbed (acid sulfate)	0.5[b] (0.2)	11.8[c] (3.2)	490[c] (370)
	Tailings	1.1[b] (0.5)	23.8[bc] (4.2)	2375[bc] (541)
	Waste rock	6.0[b] (1.0)	49.3[a] (7.2)	7050[ab] (2890)
Masonic	Unmineralized/undisturbed (andesite)	25.3[a] (4.1)	46.0[a] (7.2)	10,900[a] (592)
	Mineralized/undisturbed (acid sulfate)	0.8[c] (0.8)	13.7[b] (7.2)	24[b] (24)
	Tailings	0.8[bc] (0.8)	8.7[b] (4.4)	250[b] (144)
	Waste rock	18.2[ab] (7.9)	25.3[ab] (11.1)	1060[b] (195)

[A] ANPP: aboveground net primary productivity.
[B] Shrubs were not present in this system.

conductivity, total C, N, S, bioavailable and total metals) and physical parameters (particle size, bulk density, soil moisture).

2.4. Statistical analyses and SQI development

Data were analyzed for normality and transformed as necessary for statistical analyses at all sites; all presented data are untransformed. For each parameter (Tables 1a and 1b), one-way analysis of variance and Tukey's HSD comparisons were used to determine the minimum significant difference between the unmineralized/undisturbed, mineralized/undisturbed, waste rock, and tailings samples within a given site at a significance level (α) of 0.05.

We developed our SQI using methods adapted from Harris et al. (1996), Andrews and Carroll (2001) and Rezaei et al. (2006). Relationships between the soil and vegetation parameters (ANPP, canopy cover, and shrub density) identify a minimum data set (MDS) of soil parameters that are transformed via scoring functions to produce an SQI for each site and are described here. In the initial step, soil microbial, physical and chemical variables are selected based on a Pearson correlation coefficient > 0.50 relative to any of the vegetation parameters (Rezaei et al., 2006). To further reduce this data set, the selected variables are analyzed by principal components analysis (PCA). Only those principal components (PCs) that explained at least 5% of the variance are used for further analysis (Andrews and Carroll, 2001). Within those PCs, the highest weighted variable is selected, along with variables having an absolute value within 10% of the highest weighted variable. Multiple variables within a given PC are examined for colinearity by examining their multiple correlation coefficients. As with Andrews and Carroll (2001) and Rezaei et al. (2006), variables with correlations >0.70 are considered redundant and removed; those with the lowest correlation sums are retained in the MDS, which is generally a combination of 5–10 soil microbial, physical and chemical variables.

Variables from the MDS are then transformed for use in the SQI. We utilize indicator dependent non-linear scoring functions with a 'more is better' upper asymptote sigmoid curve, a 'less is better' lower asymptote sigmoid curve, or a midpoint optimum bell shaped curve (Karlen and Stott, 1994). All microbial variables and macronutrients are assigned 'more is better' functions; bulk density and metals such as Pb are assigned 'less is better' functions; micronutrients, pH and EC are assigned midpoint optimum functions. The y-axis values represent a normalized scoring range of 0–1 for each variable. We base the x-axis range on the mean and two standard deviations, compared to the approach used by Andrews and Carroll (2001) who set the x-axis range to within 5% of the observed range of values. Other researchers rely on expert opinion and/or published values to determine the x-axis (Harris et al., 1996; Glover et al., 2000; Masto et al., 2008). Once the variable is transformed with the scoring function, it is weighted based on the percent of variance explained from the PC in which it is located. Finally, weighted scores of all MDS variables are added together to determine the SQI. We conducted multiple regression analysis using the independent MDS variables against the dependent vegetation parameters (canopy cover, ANPP, shrub density) to validate the approach. The outcome of this approach is discussed below.

3. Results and discussion

The complete dataset, including all soil physicochemical and vegetation data, can be found in Blecker et al. (2010).

3.1. Vegetative parameters

Results for the vegetative parameters are presented in Table 2. Despite vastly different vegetation types, the undisturbed andesite and serpentine areas at Chinese Camp show similar ANPP values, though much greater canopy cover for the former. Waste rock and tailings have significantly lower shrub densities compared to the undisturbed serpentine; the latter also had lower ANPP. Similarities in canopy cover between the waste rock and serpentine soils, coupled with a 4-fold increase in buckbrush density in the latter, points to the much faster recovery of forbs and grasses relative to the shrubs in the disturbed systems. At the Battle Mountain site, the Cu/Mo porphyry soils have lower shrub densities compared to the unmineralized soils, while both the waste rock and tailing have lower values for all vegetative parameters. At both Castle Peak and Masonic study sites, the andesite soils exceed the acid sulfate soils for all vegetative parameters, the latter having similar values to the tailings piles. Vegetative parameters for the waste rock piles are generally in between the andesite and acid sulfate soils.

3.1.1. Chinese Camp (serpentine mineralization)—MDS selection and validation

Data from the serpentine soils were used to derive the MDS as the waste rock and tailings sample areas are located within the serpentine soils and underlying serpentinite bedrock. Table 3 lists the soil variables with a Pearson correlation coefficient > 0.50 compared with one or more of the vegetation parameters (ANPP, canopy cover, shrub density) for the Chinese Camp study site. The first 4 PCs explain more than 96% of the variation in potential soil quality indicators. Variables selected for the MDS (i.e. those variables within ± 10% of the highest weight within each PC) are microbial variables (acid phosphatase and mineralized-C), macronutrients (total-S) and metals (total-Mn and total-Zn). Total metal concentrations provid better correlations with vegetation parameters than bioavailable (DTPA-extractable) metals, which could indicate that DTPA is not a suitable extractant for plant-available metals in serpentine soils. Though not included in the final MDS, nitrogen (both TN and NO_3) and Total-P are identified in the initial PCA and thus could be important factors in vegetative productivity in these serpentinite systems. Of the other commonly associated edaphic limitations to vegetation in serpentinite systems (e.g., K and Ca availability, tolerance to Mg and heavy metals such as Ni, Cr, and Co; Brooks, 1987; Proctor, 1999), PCA may not be sensitive enough to identify these indicators, or the vegetation in this particular system has adapted to these stressors (Brady et al., 2005). Differences in ANPP, canopy cover, and shrub density appear to be controlled to a greater extent by S, Mn and Zn (Table 3). Results of the MLR validations are presented in Table 4, with correlations between the MDS variables and the vegetative parameters ranging from 0.64 to 0.84. Total-Mn, Total-S and mineralizable-C provid the most significant relationships with the vegetative parameters.

3.1.2. Chinese Camp (serpentine mineralization)—SQI interpretation

Fig. 1 shows the SQI scores for the PCA selected MDS variables at the Chinese Camp study site. Differences in thickness of the bar segments are related to the indicator weightings from Table 3 and reflect the relative contribution of each variable to the percent variance explained by the PCA. Acid phosphatase, mineralized-C and total-S were scored using a 'more is better' function, total-Mn and -Zn were scored using midpoint optimum functions.

The microbial variables acid phosphatase and mineralized-C account for 54% of the total SQI score for the serpentine soil. Total-S, -Mn and -Zn account for the remaining score. Though this index was developed using data from the serpentine system, it was applied to the adjacent andesite oak/grassland ecosystem for comparison. Despite the stark difference in vegetation, undisturbed andesite and serpentine soils show similar scores (1.23 and 1.16, respectively), relating the importance of these particular variables across ecosystems within close proximity to one another.

Table 3
Potential minimum data set (MDS) variables and results from principal components analysis. **Boldface numbers** represent factors with the highest loading within each principal component (PC) and those factors within 10% of the factor with the highest loading. **Boldface variables** represent those indicators selected for the SQI after redundancy analysis.

Chinese Camp	PC 1	PC 2	PC 3	PC 4
Eigenvalue	3.97	2.38	1.68	1.11
Percent variance explained	39.73	28.84	16.82	11.12
Cumulative percent	39.73	68.57	85.39	96.51
Weighting	0.41	0.30	0.17	0.12
Potential MDS variables	Eigenvectors			
AWCD[a]	0.367	−0.290	0.032	0.381
Acid phosphatase[a]	**0.455**	0.184	0.009	0.270
Mineralized-C[b]	0.321	**0.448**	−0.127	0.072
Total-S[b]	−0.256	**0.424**	0.029	−0.477
TN[a]	0.204	0.338	0.407	0.157
Nitrate[b]	−0.382	0.056	0.397	0.352
Total-Fe[a]	0.365	−0.314	0.224	−0.288
Total-Mn[c]	−0.345	0.159	−0.194	**0.562**
Total-P[b]	0.125	0.389	0.479	−0.051
Total-Zn[b]	0.184	0.331	**−0.584**	−0.005
Battle Mountain				
Eigenvalue	3.10	1.77	1.36	1.02
Percent variance explained	38.75	22.12	17.08	12.80
Cumulative percent	38.75	60.86	77.94	90.74
Weighting	0.43	0.24	0.19	0.14
Potential MDS variables	Eigenvectors			
Arylsulphatase[a]	**0.497**	0.041	0.217	−0.019
TOC[a]	**0.428**	0.356	0.223	0.287
Total-N[a,c]	**0.411**	0.446	−0.116	0.255
Total-S[c]	−0.160	**0.566**	0.036	−0.447
Total-Ca[b]	0.026	−0.093	**−0.727**	0.466
DTPA-Pb[a]	**−0.471**	0.259	0.173	0.260
DTPA-Zn[c]	−0.200	−0.144	0.546	**0.575**
Total-Cu[c]	0.335	**−0.506**	0.179	−0.192
Castle Peak				
Eigenvalue	5.05	2.46	1.90	1.58
Percent variance explained	42.1	20.5	15.8	13.2
Cumulative percent	42.10	62.62	78.43	91.58
Weighting	0.46	0.22	0.17	0.14
Potential MDS variables	Eigenvectors			
AWCD[c]	0.308	−0.004	0.434	0.048
FDA[c]	**0.391**	0.124	0.124	−0.277
Acid phosphatase[a]	0.361	0.194	−0.228	0.250
PLFA[a]	0.257	0.309	−0.424	−0.188
TN[c]	0.347	0.155	0.041	−0.370
Ammonium[c]	0.153	−0.324	0.349	0.372
DTPA-SO_4[b]	−0.058	**0.576**	0.210	0.045
Water-soluble P[a]	0.162	0.017	−0.396	**0.577**
DTPA-Na[b]	−0.102	0.406	**0.476**	0.191
DTPA-Cu[c]	0.394	−0.203	0.094	0.220
DTPA-K[c]	0.156	−0.427	0.051	−0.355
DTPA-Mn[c]	**0.439**	0.040	0.050	0.048
Masonic				
Eigenvalue	5.23	2.52	1.03	0.68
Percent variance explained	52.28	25.23	10.3	6.76
Cumulative percent	52.28	77.51	87.81	94.57
Weighting	0.62	0.30	0.12	0.08
Potential MDS variables	Eigenvectors			
Ecoplate AWCD[a]	0.305	−0.227	0.363	**0.509**
Acid phosphatase[b]	**0.374**	0.202	0.060	0.307
Ammonium[b]	**−0.368**	0.135	0.426	0.059
Total-Zn[a]	−0.174	**0.561**	0.165	0.025
DTPA-Cu[b]	0.249	0.281	**−0.523**	0.411
DTPA-Mn[c]	0.267	0.474	−0.042	0.021
DTPA-Na[c]	0.298	0.361	0.258	−0.425
DTPA-Mg[b,c]	−0.344	0.364	0.112	0.154
DTPA-SO_4[c]	0.336	−0.088	**0.544**	0.073

Pearson correlation coefficients > 0.50.
[a] ANPP.
[b] Canopy cover (%).
[c] Shrub density (plants ha^{-1}).

Table 4
Results of multiple regressions of the minimum data set (MDS) variables against the vegetation parameters as dependent variables.

Goal	r^2	Most significant MDS variable (s)	*p*-value
Chinese Camp (MDS; acid phosphatase, mineralized-C, total-S, -Mn, -Zn)			
ANPP	0.71	Total-Mn; total-S	0.19; 0.20
Canopy cover	0.64	Total-S; mineralized-C	0.03; 0.06
Shrub density	0.84	Total-Mn	0.03
Battle Mountain (MDS; arylsulfatase, TN, total-S, -Ca, -Cu, DTPA-Pb, -Zn)			
ANPP	0.78	TN; DTPA-Pb	0.02; 0.11
Canopy cover	0.60	Total-Ca	0.04
Shrub density	0.88	DTPA-Zn; -Pb; TN; total-Ca	<0.0001; 0.003; 0.004; 0.048
Castle Peak (MDS; FDA, DTPA-Na, -Mn, -SO_4, water-extractable P)			
ANPP	0.71	Water-extractable P; DTPA-Na	0.07; 0.08
Canopy cover	0.54	DTPA-Na	0.03
Shrub density	0.88	DTPA-Mn; DTPA-SO_4	0.02; 0.003
Masonic (MDS; AWCD, acid phosphatase, NH_4, DTPA-Cu, -SO_4, -Zn)			
ANPP	0.87	DTPA-Za; NH_4; acid phosphatase	0.03; 0.06; 0.09
Canopy cover	0.85	DTPA-Cu; AWCD	0.01; 0.04
Shrub density	0.85	NH_4	0.10

The undisturbed serpentine soils (SQI = 1.16) score significantly higher than the tailings (SQI = 0.27), though only slightly higher than the waste rock (SQI = 1.06). Fig. 1 shows the degree each parameter was impacted relative to the serpentine soil. Microbial activity as measured by mineralized-C and acid phosphatase is severely inhibited in the tailings piles. Based on the SQI variables, this is likely due in part to limitations in S and Zn availability and likely driven by the overall lack of soil organic matter (Blecker et al., 2010) and vegetative cover.

3.1.3. *Battle Mountain (Cu/Mo porphyry mineralization)—MDS selection and validation*

Data from both the unmineralized and Cu/Mo porphyry soils were used to derive the MDS, given the similarity in vegetation between these two areas. Table 3 lists all the soil variables with a Pearson correlation coefficient > 0.50 compared with one or more of the vegetative parameters for the Battle Mountain study site. The first 4 PCs explain approximately 90% of the variation in potential indicators. Within PC 1, TOC and TN are highly correlated (>0.70); total N was selected as it had the lower correlation sum of the two variables. Variables selected for scoring include arylsulfatase, TN, total-S, -Ca, and -Cu, and DTPA-Pb and -Zn. The occurrence of both arylsulfatase and total-S as indicators suggests the importance of S in terms of vegetative productivity in this ecosystem. Total-N, occurring in PC 1, supports the importance of N which can be a limiting nutrient in semi-arid terrestrial systems (Hooper and Johnson, 1999; Adler et al., 2006). Interestingly, total-Cu correlated with the plant factors better than DTPA-Cu, suggesting that perhaps another measure of bioavailable Cu might be more appropriate or other metals (e.g., Zn, Fe, Pb) are interfering with the DTPA-Cu chelation. Results of the MLR validations are presented in Table 4, with correlations between the MDS variables and the vegetative parameters ranging from 0.60 to 0.88. Nutrients and metals show significant correlations.

3.1.4. *Battle Mountain (Cu/Mo porphyry mineralization)—SQI interpretation*

The SQI scores for the PCA selected MDS variables for the Battle Mountain site are presented in Fig. 2. Arylsulfatase, TN, total-S, and total-Ca were scored with 'more is better' functions. DTPA-Zn and total-Cu were scored with midpoint optimum functions. DTPA-Pb was scored using a 'less is better' function, given its lack of nutritional benefits to vegetation (Adriano, 2001). Arylsulfatase activity, TN and DTPA-Pb are the greatest contributors to the SQI value (62% of the overall SQI score). Both undisturbed soils had similar scores (SQI = 1.57 unmineralized; SQI = 1.52 mineralized), with the mineralized soil showing slight decreases in TN and total-S. The similar scores between the undisturbed soils are also reflected in the ANPP and canopy cover values; however the mineralized soil has significantly lower shrub density (Table 2). Greater soil Cu concentrations may be inhibiting shrub establishment.

Soil quality index scores for the tailings and waste rock are 0.60 and 0.68, respectively. Arylsulfatase activity is absent in the

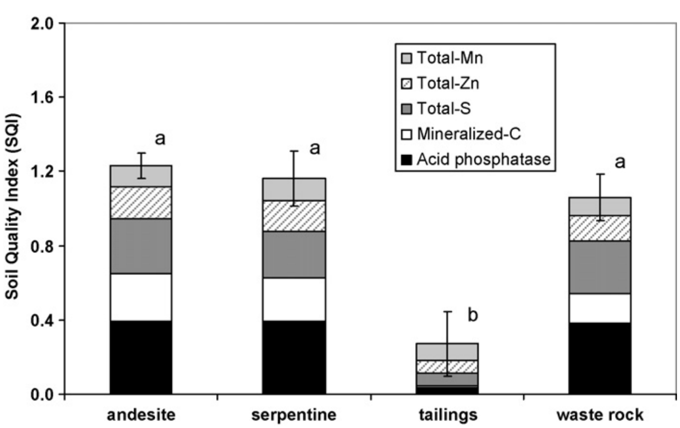

Fig. 1. Soil quality index (SQI) values for the Chinese Camp study site. Data presented are means ± 1 standard error of the mean. Different lowercase letters indicate a significant difference at $p < 0.05$.

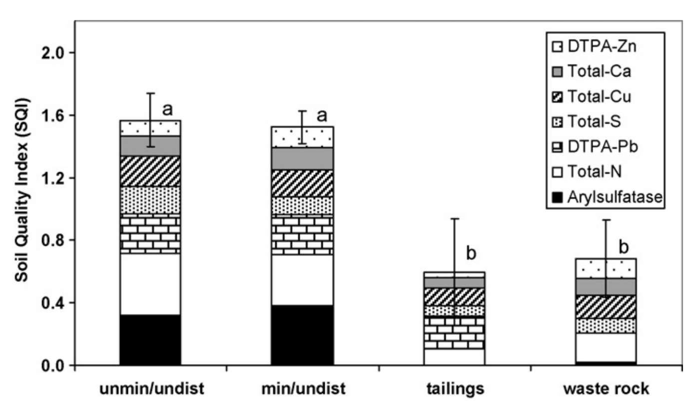

Fig. 2. Soil quality index (SQI) values for the Battle Mountain study site. Data presented are means ± 1 standard error of the mean. Different lowercase letters indicate a significant difference at $p < 0.05$.

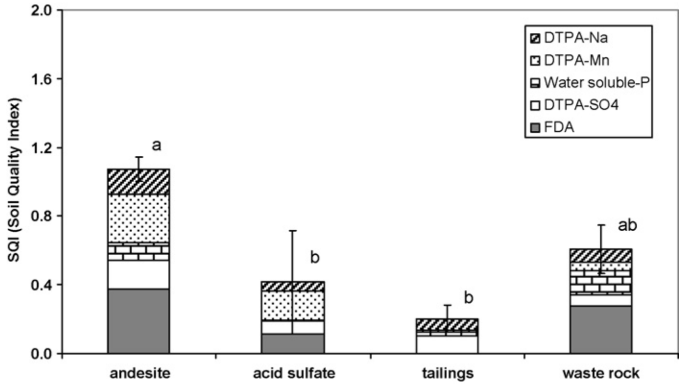

Fig. 3. Soil quality index (SQI) values for the Castle Peak study site. Data presented are means ± 1 standard error of the mean. Different lowercase letters indicate a significant difference at $p < 0.05$.

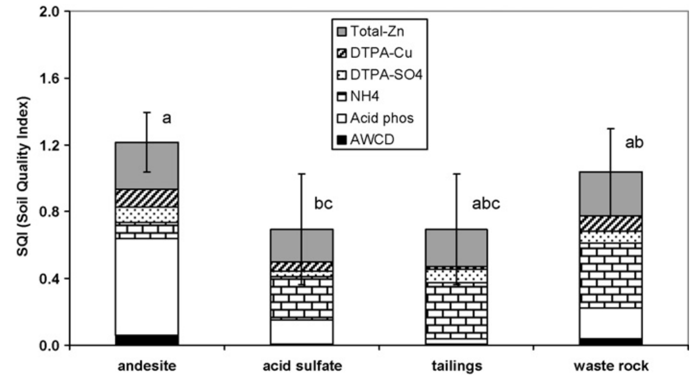

Fig. 4. Soil quality index (SQI) values for the Masonic study site. Data presented are means ± 1 standard error of the mean. Different lowercase letters indicate a significant difference at $p < 0.05$.

tailings and low in the waste rock. A combination of lower nutrient levels (e.g., TN, total-S, -Ca) and higher metals contribute to the reduced SQI scores for the tailings and waste rock areas. DTPA-Pb has little impact in the tailings (i.e., the concentrations were low), but higher concentrations in the waste rock piles contribute to the lower SQI score. Although the SQI scores are similar between the tailings and waste rock piles, the vegetation parameters are quite different (Table 2), with the waste rock showing significantly greater canopy cover and shrub density.

3.1.5. Castle Peak (acid sulfate mineralization)—MDS selection and validation

Data from the andesite soils were used to derive the MDS as the waste rock and tailings areas are located within the andesite system. Table 3 lists all the variables with a Pearson correlation coefficient > 0.50 compared with one or more of the vegetative parameters for the Castle Peak study site. The first 4 PCs explain more than 91% of the variation in potential indicators. Variables selected for scoring represent measures of microbial activity (FDA), macronutrients (DTPA-SO_4, water-soluble P), and metals (DTPA-Mn and -Na). FDA and DTPA-Mn share the highest weighting (i.e., explained the highest percentage of variance relative to the vegetation parameters). Manganese is an essential plant micronutrient, and the data suggests this may be a limiting factor in vegetative productivity in this system. The macronutrients P and S also show a link to vegetative productivity, and thus could also be limiting factors in this ecosystem. The PCA selection of Na is interesting as Na is generally not thought of as an essential plant nutrient, and the soil Na concentrations are not high enough to produce negative impacts associated with sodic soils. The positive correlation between Na and the vegetation parameters could be related to Na acting as a proxy for other cations (e.g., K, Ca, Mg), but is not known. Table 4 shows the results of the MLR's associated with the scored MDS variables and vegetative parameters ranged from $r^2 = 0.54$ to 0.88. As with previous sites shrub density is the highest correlative parameter. The macro- and micro-nutrients show the strongest relationships with the vegetative parameters.

3.1.6. Castle Peak (acid sulfate mineralization)—SQI interpretation

Fig. 3 shows the SQI scores for the PCA selected MDS variables for the Castle Peak study site. FDA, DTPA-SO_4 and water-soluble P were scored using 'more is better' functions. DTPA-Na and -Mn were scored with midpoint optimum functions. The low pH, nutrient poor acid sulfate soils score significantly lower than adjacent andesite soils, with the SQI reduced from 1.08 to 0.42. The waste rock and tailings also score lower (0.61 and 0.20, respectively). Waste rock piles show similar FDA activity and water-soluble P concentrations to the andesite soils and thus SO_4, Mn, and Na have a greater impact on the score. Tailings show little to no soil microbial activity (Fig. 3 and Blecker et al., 2010), which was likely due in part to the nutrient poor nature of these sediments. As seen in this and other studies of acid sulfate systems (DeLucia et al., 1989; Schlesinger et al., 1989), other variables not included in the SQI (such as pH, TOC, and TN) may be contributing to the vast differences in vegetative community composition and production between the acid sulfate and andesite soils.

3.1.7. Masonic (acid sulfate mineralization)—MDS selection and validation

Data from the andesite soils were used to derive the MDS as the waste rock and tailings sample areas were located within the andesite system. Table 3 lists all the variables with a Pearson correlation coefficient > 0.50 compared with one or more of the vegetative parameters for the Masonic study site (ANPP, canopy cover, shrub density). The first 4 PCs explain more than 94% of the variation in potential indicators. Variables selected for scoring represent microbial variables (acid phosphatase, AWCD), macronutrients (NH_4, DTPA-SO_4) and metals (DTPA-Zn and -Cu). The percent variance for PC 1 is considerably higher (52%) than the other PC variables (<25%), illustrating the significance of acid phosphatase activity and NH_4 in this particular ecosystem. Table 4 shows the results of the MLR's associated with the scored MDS variables, with all vegetation parameters having relatively high r^2 values (0.85–0.87). Only SO_4 lacks significant correlation with the vegetative measures, as the other 5 MDS variables are significant in one or more MLR equations. As with previous sites, macro- and micro-nutrients show the strongest relationships with the vegetative parameters.

3.1.8. Masonic (acid sulfate mineralization)—SQI interpretation

Fig. 4 shows the SQI scores for the PCA selected MDS variables for the Masonic study site. Acid phosphatase, AWCD, NH_4 and DTPA-SO_4 were scored with 'more is better' functions. Total-Zn and DTPA-Cu were scored using midpoint optimum functions. Acid phosphatase and NH_4 comprised over half of the total SQI score based on their high PCA weighting. Again, the acid sulfate soils scored significantly lower than the andesite soils (1.21 vs. 0.69, respectively), largely driven by the decreased microbial activity. Although microbial activity was inhibited in the tailings, high NH_4 concentrations and favorable DTPA-Zn concentrations led to an unexpectedly high SQI (0.69). The waste rock SQI score (1.03) was only slightly slower than the andesite soils, primarily due to reduced microbial activity. Ammonium levels were significantly higher in the waste rock relative to the surrounding andesite sites, thus other physical or biochemical factors were likely inhibiting microbial activity within the waste rock.

3.2. General discussion

Overall, SQI indicators are typically comprised of a combination of one or more measures of microbial activity and various nutrients and metals. Measures of enzyme activity are commonly included SQI variables across all study sites, and are generally less intensive than other microbial analyses. Additional measures of enzyme activity such as dehydrogenase, urease, and β-glucosidase, could provide a cost-effective means in monitoring ecosystem recovery against more expensive measures such as microbial biomass-C (PLFA) and AWCD. Nitrogen, as either TN or NH_4 is another indicator present in most SQIs, relating its important role and limiting nature in biotic activity of terrestrial ecosystems. Other macronutrients such as S and P are SQI indicators that could provide insight into the likelihood of successful reclamation efforts. For this study, indicators such as N and P were scored as "more is better" functions, but in more humid systems where runoff and aquatic eutrophication are of greater concern, could be scored as optimum functions to penalize excessive amounts (Andrews and Carroll, 2001). Even in arid and semi-arid environments, excess N can lead to undesirable changes in plant species composition (e.g., invasion by weedy species) or inhibit successional development. Somewhat surprisingly, the bioavailable (DTPA) form of micronutrients/metals such as Cu, Zn, Mn did not always correlate higher than total metals, but as mentioned previously, preference for other metals by that extractant may be interfering.

Notable parameters, such as soil moisture, pH, and TOC, found to be important in SQIs from other studies (Andrews et al., 2002; Rezaei et al., 2006; Masto et al., 2008) does not appear in the SQIs from our study. However, we do not recommend the exclusion of their measurement, as they can be invaluable in terms of explaining site or treatment differences that are not captured by an SQI. Soil moisture, in particular, is a key driver of biotic activity in these and other arid and semi-arid systems. However soil moisture does not correlate strongly with the vegetative measurements and thus was not selected as one of the SQI indicators. In most cases soil moisture is not significantly different between the different soils, and often soils with low vegetative productivity numbers show higher soil moisture content due in part to lower transpiration. In other words, sites undergoing remediation in shrubland systems can have higher soil moisture contents given the lower vegetative cover (Canton et al., 2004; Mueller et al., 2008). Another key variable, soil pH, is also absent from the SQIs. Though soil pH does not correlate with the vegetative measures, it is nonetheless a key to nutrient availability, metal speciation and toxicity.

3.3. Comparison with similar approaches used in previous studies

3.3.1. Agroecosystems

One of the earliest uses of this approach was to evaluate the impact of litter disposal treatments on agroecosystems (Andrews and Carroll, 2001). The authors identified 4–5 indicators at two sites, where SQI parameters were based on crop yield and issues of environmental concern (e.g., P runoff and As contamination) and believed this tool to be an effective means of monitoring sustainable management. Later applications in other agroecosystems showed the utility and adaptability of this approach under varied management goals (Andrews et al., 2002; Sharma et al., 2005; Masto et al., 2008).

It is important to note that a certain degree of subjectivity can exist in indicator selection and scoring when following this type of SQI approach. This isn't necessarily detrimental, but does illustrate the difficulty of standardizing indicator selection using this technique. For example, Masto et al. (2008) modified this approach by identifying soil indicators associated with six functions related to agricultural productivity, instead of relaying on the PCA-MDS indicator method. This modification is more applicable to systems with a greater body of existing initial data, where the relationships between soil parameters and vegetation parameters or management goals are better known or have been studied in more detail. It could be construed that the PCA method employed in this study may help identify those biotic/abiotic relationships in areas with little to no existing data, which was the case in the current study.

3.3.2. Other ecosystems

One of the first adaptations of this SQI technique to non-agricultural systems was undertaken by Rezaei et al. (2006) for semi-arid rangeland systems, where the SQI was based on ANPP. The authors examined spatial relationships (e.g., incorporating different slopes and aspects) of physicochemical parameters to develop a minimum data set that best described differences in ANPP. The preliminary nature of our study precluded the inclusion of that type of spatial variability, but emphasizes the utility of increasing spatial variability in order to more accurately extrapolate the results. Bastida et al. (2006) developed a microbiological degradation index for natural soils in a semiarid climate based on the Andrews and Carroll (2001) approach and found that biochemical (enzyme activity and microbial respiration) and carbon-related parameters (water soluble carbon and carbohydrates) showed the strongest relationships with plant cover density. They believed the SQI could identify a threshold, below which management action would be required to prevent further landscape degradation. Though predating Andrews and Carroll (2001), Kelting et al. (1999) used a similar approach to identify and score soil parameters in a forested system. These examples and results from the current study point out the adaptability of this approach and utility of this type of tool outside of agroecosystems, given the availability of undisturbed reference systems (Andrews and Carroll, 2001).

4. Conclusions

The SQIs at all sites were comprised of a combination of microbial activity, organic matter components, macronutrients, and micronutrients/metals; however similarities between SQI indicators across the entire study were limited. This may be due in part to the diversity in ecosystem and rock types across the study and points to the different drivers of vegetative productivity. Thus application of this approach at additional sites will be necessary to identify trends in indicator commonality. Key drivers such as soil moisture and pH are not considered here, but should prove to be important measures and explanatory variables as this SQI approach is developed. Refinements to approaches such as this also need to consider invasive species which can comprise a highly variable and sometimes significant component of vegetative communities. This SQI approach provides a means of quantifying disturbed ecosystem recovery resulting from mining and could work for other disturbances or changes in land use (e.g., land utilized for carbon sequestration) in a way that readily distills the information for potential use by land managers. However, severely disturbed areas such as unreclaimed tailings and waste rock have likely crossed an ecological threshold that precludes the use of this type of monitoring tool. Increasing the spatial gradients (e.g., slope and aspect, additional sites within a given ecoregion/geology) and temporal gradients (e.g., seasonal impacts related to moisture) beyond those used in this preliminary study are necessary to improve and increase the scale at which this method can be applied.

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