**Indicators and benchmarks for wind erosion monitoring, assessment and management**

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**Abstract**

[Text]

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**1. Introduction**

Wind erosion is a major resource concern because it impacts land health, agricultural production, ecosystem function, human health and climate (UNEP, WMO, UNCCD, 2016). Managing wind erosion and blowing dust is an urgent challenge for adapting agroecosystems to climate change and maintaining ecosystem services, such as clean air, in drylands (Webb et al., 2017a). The negative impacts of wind erosion are generally recognized within this context (Middleton et al., 2017). However, limited integrated information and crude estimates have long hampered assessments (Lal, 2001; Shao et al., 2011). These sources of uncertainty continue to affect management responses. Allocating resources to manage wind erosion is difficult where the problem is unrecognized, unquantified, and effects of management poorly understood (UNEP, WMO, UNCCD, 2016). Improved monitoring would ensure that appropriate effort is directed toward managing wind erosion, especially when other resource concerns (e.g., invasive species, habitat loss, biodiversity decline) are more readily quantified and perceived (Rodríguez et al., 2006). Because wind erosion impacts such a wide range of ecosystem services, reducing wind erosion can have multiple benefits. Approaches are therefore needed to guide wind erosion monitoring and inform management across agroecosystems.

To be effective, wind erosion monitoring requires explicit articulation ofobjectives for which monitoring information can be interpreted and translated into management actions (Lindenmayer et al., 2013; Fischman and Ruhl, 2016). ***Management objectives*** should be developed to express desired targets for changes in wind erosion and/or air quality. ***Monitoring objectives*** should establish quantitative guidelines for detecting change in targets associated with management objectives (Figure 1). Monitoring objectives define desired values, or changes in value, of indicators of wind erosion for some proportion of an assessment area and time period that should be detected at a certain confidence level relative to a benchmark. ***Indicators*** of wind erosion are variables that can be derived or directly measured in the field or by remote sensing using cost-effective and repeatable methods and are accessible to and interpretable by practitioners (land managers, agencies, and policy makers). ***Benchmarks*** are indicator values, or ranges of values, that describe desired conditions defined by management objectives and trigger the need to adjust management practices, collect additional data, or indicate management success. Identifying a core set of wind erosion indicators, methods to establish benchmarks, and design of credible systems to detect change enables practitioners to use monitoring data to make objective and decisive decisions about the effectiveness of wind erosion management, and when current management strategies should be reviewed, amended, or changed altogether.

The most coordinated approaches currently used to measure and monitor indicators of wind erosion and blowing dust are meteorological and aerosol monitoring networks. Examples include the global Aerosol Robotic Network (AERONET), US Interagency Monitoring of Protected Visual Environments (IMPROVE), Campaign on Atmospheric Aerosol Research network of China (CARE-China), and Australian DustWatch program (Leys et al., 2008). Satellite observations and numerical modeling also support these monitoring efforts and provide early warning; e.g., as part of the World Meteorological Organization’s (WMO) Sand and Dust Storm Warning Advisory and Assessment System, SDS-WAS (WMO, 2015). However, with few exceptions (e.g., Leys et al., 2009; Love et al., 2019) these networks do not address which areas are eroding, and why, with enough accuracy to inform management. Recent developments in remote sensing (e.g., Chappell et al., 2019) and monitoring of global rangelands has increased the amount of data needed to identify and characterize dust source regions, e.g., the US Bureau of Land Management’s (BLM) public lands Assessment, Inventory and Monitoring (AIM) program, Natural Resources Conservation Service’s (NRCS) private lands National Resources Inventory (NRI), Australian Terrestrial Ecosystem Research Network’s (TERN) AusPlots, and Mongolian National Agency for Meteorology and Environmental Monitoring (NAMEM). However, data collected by these programs have largely not yet been utilized to inform wind erosion assessments and management (Webb et al., 2017b). This is because the capacity to obtain indicators of wind erosion from these datasets has not always been apparent to practitioners, along with approaches to establish benchmarks to interpret monitoring data to guide management actions.

This paper identifies how indicators and benchmarks can be used to support wind erosion monitoring, assessment and management across agroecological systems. Our specific objectives are to: (1) review indicators of wind erosion and blowing dust that are currently available to practitioners; and (2) describe approaches to establishing benchmarks to support wind erosion assessments and management. We find that while numerous indicators are available for monitoring wind erosion, only a subset of these have been used routinely and most monitoring efforts have focused on air quality impacts of dust. Indicators of wind erosion collected by agroecological monitoring programs and remote sensing can provide information that more directly supports management. Establishing monitoring benchmarks is critical to formulate clear management objectives, translate them to actions, and enable management success.

**2. Available indicators of wind erosion and blowing dust**

Five types of indicators have been used by practitioners to support quantitative and qualitative assessments of wind erosion. These indicators include: 1) soil properties; 2) exposure to potentially erosive winds; 3) land health attributes based on measurements and local/expert opinion; 4) blowing dust occurrence and air quality measures; and 5) modelled sediment transport rates (Table 1). Direct measurements of aeolian sediment transport and more technical indicators of soil and site susceptibility to wind erosion (erodibility) have been developed for croplands and rangelands (see review by Webb and Strong, 2011). However, these require specialized instrumentation to measure, and can be difficult to interpret, so have not been widely adopted outside the aeolian research community (Zobeck et al., 2003). Here, we focus on indicators that are readily obtained from measurements collected by producers, resource managers, and agencies in the United States (US) and globally.

**2.1 Indicators based on soil properties**

Following the 1930s Dust Bowl, the USDA Soil Conservation Service (SCS) sought to identify properties of agricultural soils that describe their susceptibility to wind erosion. Extensive work by W.S. Chepil and colleagues through the 1940s to 60s identified soil texture (Chepil, 1953), the proportion of dry aggregates in the surface soil <0.84 mm diameter (the “erodible fraction”; Chepil, 1951), and calcium carbonate and organic matter contents (Chepil, 1954), as key indicators of erodibility that could be used to classify soils into Wind Erodibility Groups (WEGs). From this classification, a Wind Erodibility Index (“I” factor) was developed as an expression of dry soil aggregate stability under tillage and abrasion for the Wind Erosion Equation (WEQ), an empirical equation to estimate site potential soil loss by wind (t ha-1 yr-1) (Woodruff and Siddoway, 1965).

The WEGs and “I” factor are used by the USDA-NRCS (formerly the SCS) and by practitioners globally as core indicators of soil erodibility to wind (NRCS, 2018). While the classifications are often ascribed to soils based on surface texture and appear easy to apply now that texture data are available globally (e.g., Hengl et al., 2017), the WEGs and “I” factor have limitations that make them inaccurate in many soil-landscape settings and management contexts (Webb and Strong, 2011). The indicators describe the susceptibility of non-crusted soils to be mobilized by wind and do not necessarily represent their potential to emit fine dust. Soil mobility is of concern for reducing soil and nutrient losses to erosion and maintaining soil health and ecosystem function (Hagen and Lyles, 1985; Lyles and Tatarko, 1986; Danfeng et al., 2018). However, dust emission potential isn’t always directly related to soil mobility (Shao et al., 2011). Dust emission is related to the availability of fine silt and clay particles in the soil and the availability of loose sediment that can saltate and release dust particles by abrasion (Bullard et al., 2009; Kok et al., 2014). These conditions are determined by dynamic soil surface properties (e.g., crusting) and landscape context (e.g., depressions that accumulate fine sediments and are proximal to mobile sandier soils). Physical and biological soil crusts are critical for reducing the supply of erodible material and increasing soil resistance to mobilization and abrasion (Eldridge and Leys, 2003; Belnap et al., 2014). As the WEGs and “I” factor were defined for non-crusted surfaces, they do not work well for rangeland soils (Woodruff and Siddoway, 1965). As the classifications are static, they are also insensitive to land management activities that disturb soils (e.g., due to machinery, livestock and fire) and change both potential soil mobility and dust emission (Baddock et al., 2011). Because of these limitations, use of the WEGs and “I” factor on crusted soils and as indicators of wind erosion is potentially misleading. Other, more dynamic indicators of wind erosion are often more appropriate.

**2.2 Indicators of exposure to erosive winds**

The first-order control on where, when, and how much wind erosion occurs on a landscape is the amount of erosive wind energy that can act on exposed bare soil (Gillette, 1999). Surface roughness, including vegetation, rocks, gravel, embedded litter, plant residues, and soil crusts and aggregates attenuate wind erosivity by i) directly covering the soil surface; ii) absorbing a portion of the wind momentum; and iii) producing shelter areas of flow separation with reduced momentum downwind of roughness elements (Raupach et al., 1993). Surface roughness directly protects the soil surface and can be measured relatively easily in the field using standardized methods (e.g., Herrick et al., 2018).

Ground cover (as viewed from above) is a familiar concept to practitioners and has often been used as a dynamic indicator of land susceptibility to wind erosion (e.g., Webb et al., 2009; Pierre et al., 2017). National monitoring programs and the United Nations Convention to Combat Desertification (UNCCD) have adopted metrics of ground cover, collected in the field (e.g., AIM, Toevs et al., 2011) and by remote sensing (e.g., Guerschman et al., 2018; Jones et al., 2018), to monitor risk of wind erosion and land degradation (Cowie et al., 2018). In Australia, field monitoring, remote sensing and modelling are used together to assess effects of drought and land management on changing ground cover and wind erosion (Leys et al., 2009). Data availability makes the indicator attractive for local to regional wind erosion assessments. However, fractional ground cover on its own is demonstrably a poor indicator of exposure to wind erosion (Chappell et al., 2018).

Except for dust devils, wind erosion is driven primarily by lateral wind forces. Wind momentum absorption and sheltering by surface roughness therefore have the greatest effect on reducing sediment transport (Hagen and Armbrust, 1992). These processes are moderated by the vertical structure, density and spatial distribution of roughness elements, which are not described by fractional ground cover (Figure 2). Vegetation shape, porosity and flexibility (including leaf and stem area) also influence momentum absorption and sheltering (Mayaud and Webb, 2017), and have been used as indicators of erosion risk in croplands (Hagen and Armbrust, 1994; Armbrust and Bilbro, 1997). Without information on the height and distribution of ground cover, assessments may severely over- or underestimate erosion risk (Webb et al., 2014a). Alternatively, indicators of vegetation canopy height and canopy gap size distribution (measured as the spacing between plant canopies) can be collected in the field using standardized methods (Herrick et al., 2018). Together, these structural indicators can be used to explain to the first-order where wind erosion may occur (Okin, 2008).

Indicators of vegetation structure have been measured extensively across western US rangelands (NRI, Goebel, 1998; AIM, Toevs et al., 2011), Mongolian grasslands (NAMEM, Densambuu et al., 2018), and at select cropland sites (Webb et al., 2016), and used to assess wind erosion across plot (< 1 ha) to regional (>106 ha) scales (e.g., RCA, 2011). The indicators can be integrated to estimate sediment transport using available models (Vest et al., 2013; Webb et al., 2014b), although canopy gap distribution information has generally not been included in wind erosion models for croplands (e.g., Tatarko et al., 2016). Recent remote sensing advances have enabled area-integrated measures of surface roughness that approximate surface protection by aerodynamic sheltering from the shadow cast by roughness elements (Chappell et al., 2010; Chappell and Webb, 2016). The approach enables global wind erosion estimates at a moderate (500 m) spatial resolution across land cover types, filling gaps in field monitoring (Chappell et al., 2019). Vegetation structural properties can be related to land health attributes that provide complementary qualitative indicators of wind erosion.

**2.3 Indicators of land health attributes including soil properties and exposure**

Several global initiatives to assess land health include indicators of wind erosion (e.g., FAO, 2010; WOCAT, 2010; International Resource Panel, 2016; Cowie et al., 2018). Of these, the Interpreting Indicators of Rangeland Health (IIRH) assessment protocol (Pyke et al., 2002; Pellant et al., 2005) has been applied at over 30,000 sites in the US and integrates 17 indicators of rangeland health into three attributes of soil and site stability, hydrological function and biotic integrity (Herrick et al., 2019). Quantitative and qualitative indicators of wind erosion are incorporated into IIRH assessments, including the presence of pedestals and/or terracettes, bare ground cover, presence of wind scouring, blowouts and/or sediment deposition areas, litter movement, and soil surface horizon loss or degradation. Related protocols like Landscape Function Analysis (Tongway and Hindley, 2004) and Pedoderm and Pattern Class (Burkett et al., 2013) consider similar indicators.

Qualitative indicators of soil and site stability and biotic integrity provide information on the status of wind erosion at a site that may not be obtained from quantitative indicators of soil erodibility and site exposure to erosive winds. Interpreting qualitative indicators like those used in IIRH requires a descriptive reference of the status of the indicators for a defined “healthy” site (one with a minimal departure from reference conditions), or classification for rating indicators that enables practitioners to define relative expected values for a historical reference (Herrick et al., 2019). This approach has the benefit of incorporating reference benchmarks into assessments that can help diagnose whether wind erosion is a problem, in addition to providing another line of evidence in support of more quantitative monitoring and modeling. However, wind erosion should not be assessed using land health attributes alone as they are telling of land status, influenced by past management, and not necessarily current or potential erosion. Wind erosion and dust emission that functionally impact land health and degrade air quality may occur, or may have recently occurred, without visual evidence to suggest that land is departing from the reference condition.

**2.4 Indicators of atmospheric dust and air quality**

Dust monitoring globally has been coordinated through meteorological observation networks and aerosol measurement networks (Goudie and Middleton, 2006). Indicators used by these networks include: dust event frequencies obtained from visual observations (e.g., McTainsh et al., 1998; Shao and Dong, 2006; O’Loingsigh et al., 2010, 2014); atmospheric particulate matter (PM) concentrations measured using high volume air samplers, lidar, and light-scattering laser photometers (e.g., Xin et al., 2015; Hand et al., 2016; Love et al., 2019); and aerosol optical depth (AOD) obtained from ground-based sun photometers and satellite observations (e.g., Holben et al., 2001; Prospero et al., 2002; Ginoux et al., 2012). While these indicators directly relate to climate, visibility and human health impacts of dust, attributing transported dust loads to specific upwind source areas and land uses with enough accuracy to inform management is very challenging (McTainsh et al., 1998; Webb and Pierre, 2018). Additionally, dust aerosol data are generally collected and interpreted by agencies, or divisions of agencies, that have interests in air quality (e.g., environmental protection agencies) but may have little or no formal connection to programs responsible for monitoring and managing source area soils and vegetation. There is therefore an opportunity to extend the benefits from monitoring dust by creating or strengthening data and knowledge sharing with practitioners who have a stake in managing eroding landscapes (e.g., through USDA-NRCS’s National Air Quality Initiative). The New South Wales Office of Environment and Heritage provides an example of such integration – collecting and publishing hourly aerosol data on the Rural Air Quality Network (<https://www.environment.nsw.gov.au/topics/air/monitoring-air-quality/regional-and-rural-nsw/rural-monitoring-stations/live-air-quality-data>). Air quality data are then interpreted in conjunction with remotely sensed ground cover, rainfall and fire data to report on causes of wind erosion in south-eastern Australia via the Community DustWatch project (Leys et al., 2008).

**3. Indicators for multiple management objectives**

When selecting indicators of potential wind erosion, it is important to consider that management decisions about wind erosion are rarely made in isolation of other conservation and production objectives. To reduce the need for costly dedicated monitoring, wind erosion indicators would therefore ideally also be appropriate for assessing the status, condition, and trend of other ecosystem services of management interest; that is, are multi-use and available from or have value to existing monitoring programs (Probst and Stelzenmüller, 2015). Fortunately, indicators derived from a core set of standardized measurements that are widely used to monitor land health, invasive species, habitat quality and production can also be used to monitor and assess wind erosion. These indicators include surface soil texture, vegetation and other ground cover (by species/type), vegetation canopy height, and canopy gap size distribution (Herrick et al., 2018), and can be supported by related indicators obtained using remote sensing (e.g., Chappell et al., 2018; Jones et al., 2018). In the US, plot-scale measurements of surface soil texture, ground cover, vegetation canopy height, and canopy gap size distribution are collected by the AIM and NRI programs and National Wind Erosion Research Network because of their broad utility for assessing different aspects of land health across rangelands and croplands (Goebel, 1998; Toevs et al., 2011; Webb et al., 2016).

Wind erosion models can be used to integrate core indicators to support wind erosion assessments at the farm scale (e.g., Tatarko et al., 2016; Pierre et al., 2017), at plot-to-regional scales using plot monitoring data (e.g., Munson et al., 2011) and at landscape-to-global scales using remote sensing (e.g., Chappell et al., 2019). However, available models estimate soil loss (t ha-1 yr-1), or sediment transport rates (g m-1 s-1) and dust emission (g m-2 s-1); metrics that remain generally unfamiliar to practitioners as they are difficult to interpret without a defined reference. Establishing quantitative benchmarks is needed for practitioners to understand how soil and vegetation indicators relate to sediment transport and erosion rates across scales, and to assess whether sites have an acceptable, or unacceptable, risk of erosion and act accordingly (Pretorius and Cooks, 1989).

**4. Establishing benchmarks for monitoring objectives**

Benchmarks are needed to determine if observed indicator values at assessed locations are within the range of desired conditions and meet monitoring objectives. If monitoring information shows that an insufficient amount of a management area has met a benchmark, then changes in management actions can be triggered (Lindenmayer et al., 2013). Conversely, failure to set benchmarks can make it difficult to interpret monitoring data as observed values cannot be used to assess condition or the attainment of management objectives (Fischman and Ruhl, 2016).

Conceptually, wind erosion and dust emission are a function of site potential and the ecological state of a site. Site potential is defined by climoedaphic properties and determines the capacity of a site to produce certain kinds and amounts of vegetation, and its responses to disturbances and management (ref). Vegetation dynamics determine the ecological states of a site as plant communities (and dynamic soil properties) respond to biotic (e.g., competition, facilitation) and abiotic (e.g., management, disturbances) feedback mechanisms and threshold changes in processes like erosion (Bestelmeyer et al., 2003). Ecological states can be described by structural indicators such as plant community composition, ground cover, canopy gap sizes and canopy height. Persistent state changes may occur when structural thresholds are crossed (e.g., grassland to barren soil), and so structural thresholds can be identified as benchmarks to maintain land in a certain state/avoid state change or to guide restoration (Bestelmeyer, 2006). The structure of states also impacts biotic and abiotic function, with functional thresholds determining, for example, how sites are impacted by erosion (Okin et al., 2008) and how vegetation structures moderate sediment transport rates (Sasaki et al., 2018). Using structural and functional thresholds to define benchmarks focuses wind erosion management on ecosystem services and function, the mechanics of the erosion process, and its various impacts.

Structural and functional thresholds are often related but can also vary in predictability among contexts. For example, vegetation change in some sites may yield little impact on wind erosion because erosion is controlled by static attributes of the site (like gravel cover). At other sites, there may be a nonlinear change in erosion where structural thresholds vary depending on the nature of vegetation change (e.g., shrub invasion of desert grasslands). Functional thresholds for wind erosion are typically defined by the process mechanics, with sediment transport and dust emission responding nonlinearly to changes in surface aerodynamic roughness and soil erodibility (ref). Functional thresholds can also describe human health and social impacts of blowing dust; for example, for respiratory health and degradation of viewsheds.

In practice, differences in site potential can be accounted for by grouping sites based on land classification systems, like ecoregions, and using Ecological Site concepts with associated state-and-transition models to describe vegetation-management interactions (USDA, 2013; Bestelmeyer et al., 2015; Webb and Pierre, 2018). Approaches to establishing benchmarks for structural and functional thresholds of concern may draw on: i) experimental studies reported in peer-reviewed literature; ii) physically-based and empirical models; and iii) distributions of indicator values obtained from reference sites in existing monitoring data. These approaches vary in their potential for bias, our ability to quantify bias, ease of communication, applicability to management questions, and availability in the geographic region of interest (Keiter, 2004).

**4.1 Benchmarks from scientific literature**

Monitoring benchmarks should be established based on the best available science and data, and whenever possible be supported by published experimental studies. Field and laboratory wind tunnel studies addressing effects of soil properties (e.g., Chepil, 1944; Fryrear, 1985; Gillette et al., 1980) and ground cover levels (e.g., Wasson and Nanninga, 1986; Leys, 1991) on functional thresholds for wind erosion have been conducted to inform management. Using published research to establish benchmarks can add rigor to approaches that derive benchmarks from measured indicator values (Section 4.3) by informing how wind erosion responds functionally to changes in its controlling factors, and how agricultural, ecological and human systems respond to wind erosion at different levels (e.g., Okin et al., 2006). Care should be taken to ensure that literature used to inform benchmarks is rigorous and relevant to the soils and vegetation at the geographic location of interest (i.e., site potential).

**4.2 Model-informed benchmarks**

Field data from a network of monitoring sites and geospatial data can be used to model the physical relations among indicators and estimate wind erosion and dust emission. Physically-based models can be used to identify functional thresholds of concern for sediment transport that may be generalizable across sites and ecological states. For example, Webb et al., (2014b) identified functional thresholds of ground cover and vegetation canopy gap sizes for aeolian transport that were consistent across Chihuahuan Desert ecosystems. A major limitation of current wind erosion models is that, with exception of the Wind Erosion Prediction System (WEPS, Hagen, 1991), they currently do not account for dynamic soil surface properties like crusting and aggregation (Webb and Strong, 2011). Where important, other indicators of these properties should be considered alongside model estimates to enable a more complete assessment of whether management objectives are being met.

Empirical modeling can also be used to inform benchmarks by identifying or predicting reference areas of similar ecological potential (e.g., Hawkins et al., 2009; Nauman and Duniway, 2016; Nauman et al., 2017). This can be particularly useful for establishing benchmarks for structural thresholds of concern at intensively managed sites and in fields or regions where site potential or historical conditions are unknown (e.g., Bastin et al., 2012; Guerschman et al., 2018). A benefit of both physically-based and empirical modeling is that the approaches can provide quantitative error/uncertainty estimates that can be considered in management decision making. A caveat is that accuracy varies widely among available wind erosion models (Shao et al., 2011).

**4.3 Benchmarks based on desired or reference conditions**

In rangelands, reference conditions can be defined for a site, where ‘reference’ refers to areas where ecological processes are functioning (as inferred from structural and functional indicators) and/or anthropogenic disturbances are below thresholds thought to impact agroecosystem structure and function. Reference sites have been defined as locations in a ‘minimally-disturbed condition’, ‘historical condition’, ‘least-disturbed condition’ and ‘best attainable condition’ (Stoddard et al., 2016). However, the condition of ‘reference sites’ defined in any of these ways can vary through space and time as human impacts are disproportionately distributed, change through time, and can have differing impacts under certain physiographic conditions (e.g., Pickup et al 1998; Vanacker et al., 2007; Bastin et al., 2012). In addition, the use of a ‘reference’ to define benchmark conditions can be highly subjective and difficult to gain agreement among stakeholders. Similarly, criteria used to identify least-disturbed conditions can vary among indicators. The end goal of management may not be to attain reference condition, but to assess the degree of departure from the reference state and decide whether such departures are sustainable given management objectives or if a new “improved” condition should be the target. Structural thresholds (e.g., shrub density levels) can be used as benchmarks to assess departure from reference. For croplands and other intensively managed sites, reference conditions may focus on indicators of soil health (e.g., soil organic carbon, electrical conductivity, microbial biomass) or analogous rangeland sites.

***4.3.1 Use of ‘historical’ reference site networks***

Data collected at networks of reference sites can be used to develop frequency distributions of reference site indicator values and identify structural thresholds of concern. The distributions of indicator values are a characterization of the ‘historical’ range of variability expected to occur in a region in the absence of certain anthropogenic impacts. Percentiles of the resulting distributions can be used to set benchmarks, e.g., for reference ecological states, against which monitoring data can be compared and deviations from reference conditions identified (Figure 3). Reference site networks are typically grouped by categorical variables such as physiographic boundaries (e.g., ecoregions) to account for differences in reference site potential and subsequent variability resulting from factors such as climate and topography.

For example, the 90th and 70th percentiles of reference site dust emission rates for an ecoregion can be used as benchmarks to classify the condition of a monitoring site as having “major”, “moderate”, or “minimal” departure from reference conditions, respectively. In other words, a site would be categorized as having major departure from reference conditions if the dust emission rate for a site is greater than that observed among 90% of reference sites in the ecoregion. In contrast, the site would be categorized as moderate departure if the site is less than 90% of reference sites but greater than 70% and minimal departure if less than 70% of reference sites. Using this approach without accounting for natural environmental gradients within physiographic boundaries (e.g., ecoregions), can lead to over- or under-protection of sites due to inaccuracies in the benchmarks. The approach is also strongly dependent on sample size for which the indicators were measured and should represent the spatial variability of the indicators. Where possible, wind erosion benchmarks obtained from distributions of indicator values should be related to functional thresholds; e.g., ground cover over the management area that reduces dust emission below its benchmark level (Leys et al., 2018).

***4.3.2 Current conditions from other monitoring data***

In the absence of ‘historical’ reference sites, existing monitoring data can be used to establish benchmarks based on sites in an identified ‘least-disturbed’ or ‘best attainable’ condition. This is often the most achievable way to set benchmarks because finding historical reference sites is difficult in some agroecological regions and may not exist in croplands. For rangelands, a set of sites can be selected that provide a sample of locations with specific attributes (e.g., burned vs. unburned; grazed vs. ungrazed) to identify least disturbed or best attainable conditions and ensure there is sound reasoning to expect that sites represent a management objective. The fraction of sites likely in the desired condition should then be identified, considering the monitoring design used to select the sites and any site screening. Benchmarks will correspond to the indicator values of the selected sites in the desired condition. For example, the 75th percentile of dust concentration across sites could be a benchmark to differentiate acceptable and unacceptable health risk. Where possible, the quantile that is used should be informed by functional thresholds for how the indicators control erosion and impact e.g., human health, and the proportion of the management area that is desired to meet the benchmark. This approach to establishing benchmarks requires discretion as inclusion of too many degraded sites will lower benchmarks and under-protect assessed sites. Monitoring sites used to establish benchmarks should be independent of those being assessed to avoid introduction of circular reasoning into management decisions.

**5. Establishing benchmarks for policy and regulation**

Most monitoring benchmarks relating to wind erosion have been set for policy and environmental regulations for air quality. The US Clean Air Act of 1970 and Environmental Protection Agency’s (EPA) 1999 Regional Haze Rule are examples of regulatory actions to manage air quality and visibility impairment, including impacts of mineral dust. Similar regulations are in place at national, state and county levels globally and typically stipulate concentrations limits (e.g., for wilderness areas and around cities and towns) as functional thresholds for particulate matter with aerodynamic diameter <10 μm (PM10) and <2.5 μm (PM2.5) that may be suspended in the atmosphere for long periods (e.g., hours to weeks) and impact human health. For example, the US National Ambient Air Quality Standards (NAAQS) state an averaging time, level and form for each indicator – e.g., the 24-hour average PM10 concentration must not exceed 150 μg m-3 more than once per year on average over three years to meet the NAAQS (EPA, 1997).

Policy and regulations may stipulate benchmarks, and so should be referenced by managers, but they are not the mechanism used to develop a given benchmark. Benchmarks adopted as policy should be developed upon scientific understanding and identified from one or more information sources (e.g., scientific literature, models, monitoring data) on the impacts of indicator conditions on the environment and human health and safety. Science-policy briefs may also outline principles and approaches to establish benchmarks. For example, the UNCCD developed ‘Target-Setting Building Blocks’ to provide guidance for identifying indicators, setting benchmarks, and assessing progress toward Land Degradation Neutrality (LDN) that are relevant to managing wind erosion and other sustainability challenges (UNCCD, 2016).

To provide managers with flexibility in meeting targets and avoid unintended consequences, indicators of outcomes (e.g., dust emitted) should be prioritized for policy over factors that are correlated with these outcomes (e.g., vegetation cover or soil erodibility) so that managers can get the greatest return on investment from mitigation strategies for their land. Where indicators of outcomes are too expensive to measure, management flexibility may be promoted by identifying alternative indicators that may be used to predict achievement of the outcome-based targets, provided that the outcome remains the overall objective. This both helps promote innovation as managers focus on the desired result, rather than the regulation, and allows for the introduction of new less expensive measurement technologies in the future.

**6. Best practices for establishing monitoring benchmarks**

In all cases, best professional judgement based on relevant science and data should be exercised when establishing monitoring benchmarks. The approaches described above are not mutually exclusive in principle or practice. Using a combination of approaches is therefore recommended to consider and provide multiple lines of evidence, include different monitoring indicators, and ensure benchmarks are appropriate to aeolian process mechanics and relevant to broader land management objectives at relevant scales of management. Establishing benchmarks based on professional judgement alone can have risks; for example, where local opinion about reference sites, thresholds, or management objectives diverge from established literature or monitoring data (Gordon et al., 2016). A basic set of principles for developing monitoring benchmarks should therefore be followed.

To encourage rigor in benchmark development, approaches should account for differences in site potential across landscapes and thresholds of structural and/or functional concern. Air quality benchmarks may reflect blowing dust impacts but should be related to soil and vegetation controls in eroding landscapes to directly inform management. Benchmarks should therefore be established for areas defined by climoedaphic groups (e.g., Ecological Site Descriptions; USDA, 2013) using information that is relevant to the geographic area of interest. Linking wind erosion indicators to ecological sites through applications like the Ecosystem Dynamics Interpretive Tool (EDIT; Bestelmeyer et al., 2016) would improve our ability to set benchmarks, assess whether sites meet (or fail to meet) monitoring objectives, and enable wind erosion to be considered alongside other ecosystem processes and services (e.g., Galloza et al., 2018).

The currency, depth and quality of information and compatibility of data collection methods used to group sites and establish benchmarks should always be considered. This includes policy documents, regulations and scientific literature, which should be cited when benchmarks are documented with a rationale for including (and excluding) different information sources or studies. It is important to understand the geographic location and sample size of data used to establish benchmarks and how reference conditions (e.g., historical, least-disturbed, minimally disturbed) were defined and used to develop indicator distributions and models. Understanding how model uncertainties and the shape and bias of indicator distributions can affect benchmarks is important to determine whether, for example, model-informed benchmarks will result in more or less protection of resources than other data-driven approaches. All approaches for setting benchmarks are subject to error. An important part of developing and applying benchmarks is therefore to be aware of the potential limitations of the different approaches and benchmarks should be updated as new information becomes available. Overall, robust benchmarks should enable managers to assess the degree or risk of departure of sites from desired conditions and make objective and decisive decisions to maintain agroecosystem structure and function.

**7. Conclusions**

Identifying indicators and establishing benchmarks for monitoring wind erosion and blowing dust is important for wind erosion assessment and management (UNEP, WMO, UNCCD, 2016). Numerous indicators of wind erosion are routinely collected by coordinated monitoring programs, including indicators based on soil properties and vegetation characteristics, indicators of land health attributes, and indicators of air quality. Some of these (e.g., air quality indicators) are widely used among practitioners interested in the impacts of blowing dust, while available soil and vegetation datasets have been used extensively in some areas to monitor wind erosion to inform assessments and management but are currently underutilized in others.

Using monitoring data to inform wind erosion assessments and management requires indicator benchmarks that describe desired conditions and can trigger adjustments to management practices, additional data collection, or indicate management success. Wind erosion and air quality benchmarks have been established by policy and environmental regulations, and from peer-reviewed literature, but alternative approaches that use reference sites and draw on measured distributions and modelled interactions among soil and vegetation indicators in existing monitoring datasets can also be used. The best approach to establishing benchmarks is one that is based on rigorous science and/or monitoring data and documents the rationale for selecting indicator values or ranges based on functional thresholds for their environmental and human impacts. By prioritizing indicators of outcomes, management flexibility can be promoted for meeting benchmarks to avoid trade-offs and unintended consequences. The approaches to establishing benchmarks described here have broad utility for managing agroecological systems and considering co-benefits of resource management among multiple ecosystem services.

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|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Indicator** | **Description** | **Measurement approaches** | **Data availability** | **Ease of interpretation** | **Example protocols** | |
| ***Soil erodibility indicators*** | | | | | |
| Soil texture | Particle size distribution of soil, classified into size bins or classes | Hand texture analysis  Hydrometer  Laser diffraction | High | High | Bouyoucos (1951), Zobeck (2004) | |
| Dry aggregate size distribution (DASD) | Size distribution of soil aggregates. The fraction <0.84 mm is considered erodible | Rotary sieving  Hand nest sieving  Elutriation | Low | Moderate | Chepil (1952), Fryrear (1985), Lyles et al. (1970) | |
| Wind erodibility group (WEG) | Classification of soil erodibility based on soil texture class, DASD and CaCO3 content | Applied to soil texture classes | High | High | Chepil (1954) | |
| Soil surface roughness | The physical roughness of the soil surface, including random and oriented roughness (e.g., due to tillage tools) | Pin profiler  Laser profiler  Chain set method  Remote sensing | Low | Low | Potter et al. (1990), Saleh (1993), Chappell et al. (2010) | |
| Crust modulus of rupture | Shear stress required to fracture soil crust | Penetrometer  Torvane | Low | Low | Chepil and Woodruff (1963), Belnap and Gillette (1998) | |
| Threshold friction velocity (*u*\*t) | Wind shear (friction) velocity at which grains (aggregates) are mobilized by wind | Anemometers with saltation particle counter | Low | Low | Barchyn and Hugenholtz (2011) | |
| Surface crust cover | Fraction of soil surface covered by physical and/or biological soil crusts | Line-point intercept  Step point transects | High | High | Herrick et al. (2018) | |
| Loose erodible material | Fraction of loose sediment lying on the soil surface that may be mobilized by wind | Line-point intercept  Step point transects  Physical sample collection | Low | High | Herrick et al. (2018)  Zobeck (1989) | |
| ***Surface sheltering and protection indicators*** | | | | | |
| Ground cover | Surface covered by rooted plant material, rock fragments, gravel (>2 mm), embedded and loose woody and herbaceous litter | Line-point intercept  Step point  High- to moderate-resolution remote sensing | High | High | Herrick et al. (2018)  Herrick et al. (2005) | |
| Vegetation foliar cover | Fraction of surface covered by rooted woody and/or herbaceous plant material | Line-point intercept  Step point  High- to moderate-resolution remote sensing | High | High | Herrick et al. (2018) | |
| Vegetation composition | Plant species present at a site | Line-point intercept | High | High | Herrick et al. (2018) | |
| Vegetation height | Height of the tallest plant parts within site, provides vertical structure | Line-point intercept | High | High | Herrick et al. (2018) | |
| Vegetation foliar density | Leaf area index (LAI)  Stem area index (SAI) | Plant area meter  Calculation from stem diameter, height and population | Low | Low | van Donk and Skidmore (2001) | |
| Canopy gap size distribution | Size of spaces between plant canopies, describes spatial distribution of roughness | Gap intercept  High-resolution remote sensing | High | High | Herrick et al. (2018) | |
| Effective sheltering | Effective aerodynamic sheltering of soil surface by roughness elements | High- to moderate-resolution remote sensing | Moderate | Moderate | Marticorena et al. (2006), Cho et al. (2012), Chappell and Webb (2016), Potter et al. (1990) | |

Table 1 – *continued*.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Indicator** | **Description** | **Measurement approaches** | **Data availability** | **Ease of interpretation** | **Example references** | |
| ***Indicators of land health attributes*** | | | | | |
| Pedestals and/or terracettes | Plants or rocks appear elevated because of soil loss around them | Visual estimate | High | High | Pellant et al. (2005) | |
| Wind scouring, blowouts and/or sediment deposition areas | Soil loss or gains around plants, rocks and within plant interspaces | Visual estimate | High | High | Pellant et al. (2005) | |
| Soil surface horizon loss or degradation | Soil horizon(s) no longer present or loss of organic material from surface horizon | Visual estimate | High | Moderate | Pellant et al. (2005) | |
| ***Sediment transport and air quality indicators*** | | | | | |
| Saltation mass flux (*Q*) | Transport rate of sediment moving horizontally along the soil surface (g m-1 s-1) | Slot samplers  Big Spring Number Eight (BSNE)  Modified Wilson and Cooke (MWAC) | Low | Low | Fryrear (1986), Webb et al. (2016) | |
| Dust mass flux (*F*) | Vertical dust emission rate (g m-2 s-1) | Derived from vertical profile of PM concentration measurements | Low | Low | Gillette et al. | |
| Dust event frequency | Occurrence of blowing dust events, including dust storms, dust hazes, and dust devils | Visual observation for weather records | High | High | O’Loingsigh et al. (2010) | |
| Particulate matter concentration (PM) | Abundance of mineral dust particles in a volume of air (μg m-3) | Light scattering laser photometer  Lidar | High | Moderate | Hand et al. (2016) | |
| Aerosol optical depth (AOD) | Natural logarithm of the ratio of incident to transmitted radiance through the atmosphere | Sun photometer  Moderate-resolution remote sensing | High | Low | Holben et al. (2001), Green et al. (2009) | |

Figure 1 – Illustration of how measurements, indicators and benchmarks are defined by monitoring and management objectives and are used to inform management decisions. Monitoring objectives define how changes in condition, trend, or treatment effectiveness should be assessed for each management objective. Monitoring objectives include information about the indicators, benchmarks, proportion of the study area, confidence level and time over which an assessment is being made. Benchmarks should be defined for indicators identified in the monitoring objectives and which can be obtained from measurements or models.

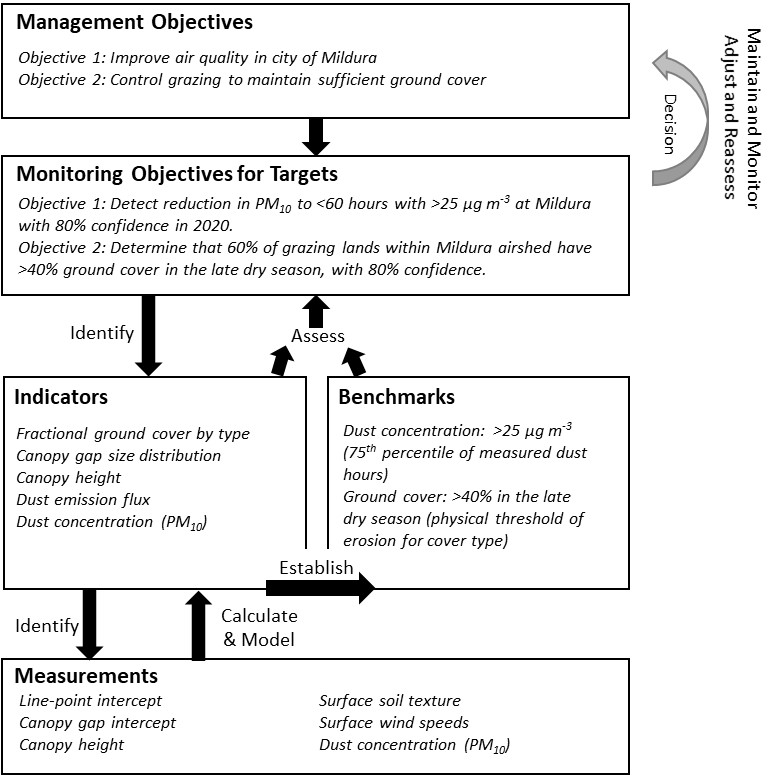


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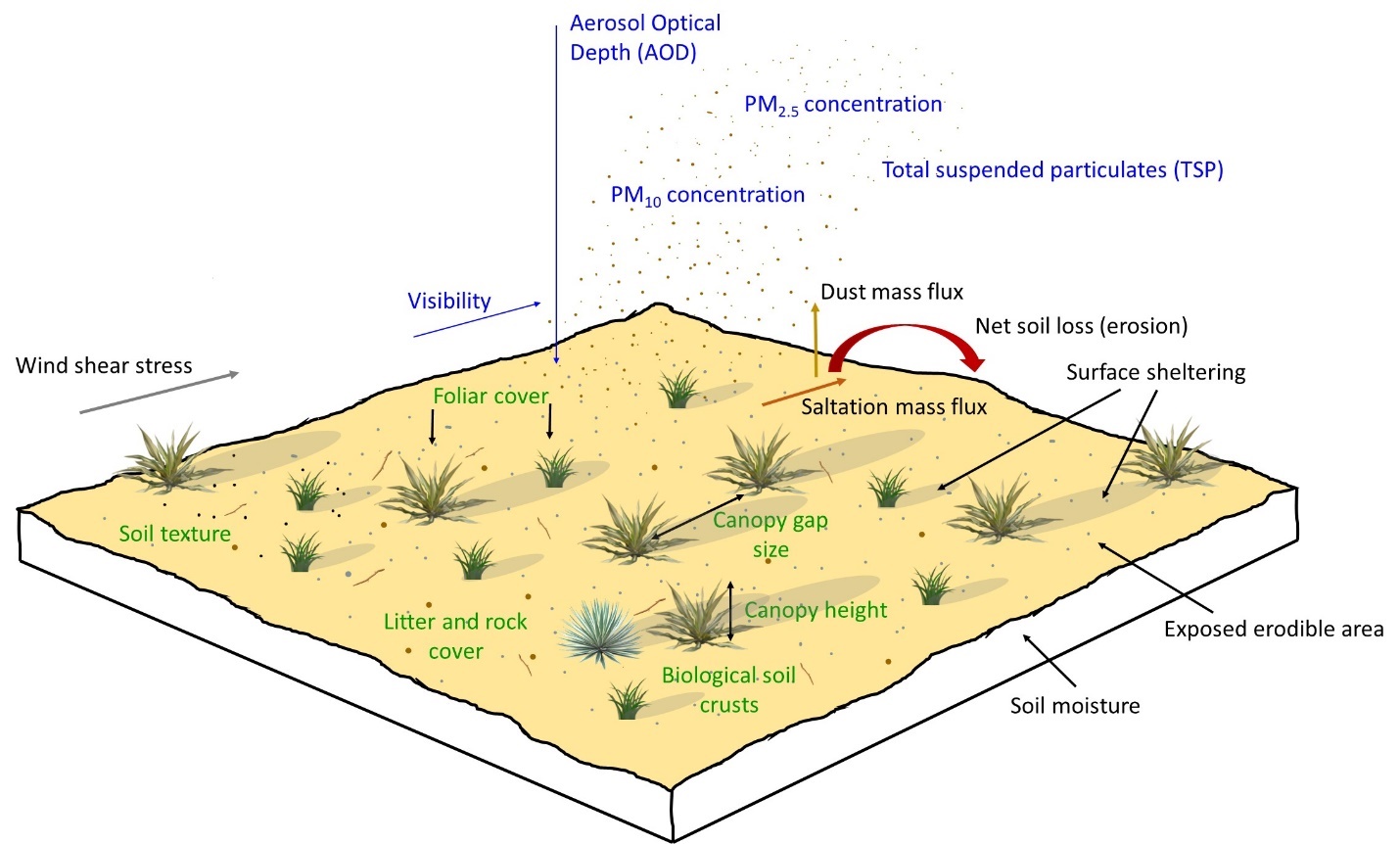


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