Loss of soil and PM10 from agricultural fields associated with high winds on the Columbia Plateau

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Abstract

Winter wheat–summer fallow is the conventional cropping system employed on >1·5 million ha within the Columbia Plateau of eastern Washington and northern Oregon. Wind erosion contributes to poor air quality in the region, yet little is known concerning the magnitude of soil and PM10 (particulate matter of ≤10 μm in aerodynamic diameter) loss from agricultural lands. Therefore, loss of soil and PM10 was assessed from a silt loam in eastern Washington during 2003 and 2004. Field sites were maintained in fallow using conventional tillage practices in 2003 (9 ha field) and 2004 (16 ha field) and instrumented to assess horizontal soil flux and PM10 concentrations at the windward and leeward positions in the field during high-wind events. Soil flux was measured using creep and airborne sediment collectors while PM10 concentrations were measured using high-volume PM10 samplers. Aggregate size distribution of parent soil and eroded sediment was characterized by rotary and sonic sieving. Six high-wind events occurred over the two year period, with soil loss ranging from 43 kg ha^{-1} for the 12–22 September 2003 event to 2320 kg ha^{-1} for the 27–29 October 2003 event. Suspension-sized particulates (<100 μm in diameter) comprised ≥90 per cent of the eroded sediment, indicating that direct suspension may be an important process by which the silt loam eroded. The corresponding loss of PM10 for these two events ranged from 5 to 210 kg ha^{-1}. Loss of PM10 comprised 9–12 per cent of the total soil loss for the six events. This study suggests that the relatively small loss of PM10 from eroding agricultural fields maintained in summer fallow can affect air quality in the Columbia Plateau. Therefore, alternative tillage practices or cropping systems are needed for minimizing PM10 emissions and improving air quality in the region. Copyright © 2006 John Wiley & Sons, Ltd.

Keywords: wind erosion; PM10; windblown dust; air quality; Columbia Plateau

Introduction

The economic and social impact of dust storms during the 1930s in the Great Plains region of the United States led to federal intervention to more widely promote the development and implementation of soil conservation practices on agricultural lands to reduce wind erosion (Lockeretz, 1978). These practices were designed to maintain or improve soil productivity for future generations. Indeed, wind erosion threatens soil productivity as a result of removing the finer particulate matter from the soil surface. More recently, fine particulates eroded from the soil surface and subsequently suspended in the atmosphere have created air quality concerns in the western United States. Saxton (1995), for example, reported that wind erosion was a major cause of non-compliance of the US EPA National Ambient Air Quality Standard for PM10 within the Columbia Plateau region of the Pacific Northwest. The semiarid climate, occasional high winds, fragile and fine-textured soils and large extent of land managed in conventional winter wheat–summer fallow (>1·5 million ha^{-1}) promote wind erosion, which contributes toward poor air quality in this region.

Wind erosion and air quality studies conducted in the Columbia Plateau region over the past decade have focused on ascertaining the magnitude of soil and PM10 transport across agricultural landscapes and developing models capable of simulating wind erosion and emission of PM10 from soils (Kjelgaard *et al.*, 2004b; Saxton *et al.*, 2000).

The wind erosion and emissions algorithm attempts to predict vertical PM10 flux as a function of the estimated horizontal soil transport. Although the aim of these previous studies was to ascertain horizontal soil transport and vertical PM10 flux from large fields managed primarily in a winter wheat–summer fallow rotation, the experimental design lacked the rigor to assess the loss of soil and PM10 from agricultural fields.

Farm conservation programs such as the USDA Conservation Reserve Program are designed to minimize the impact of farming systems on the environment. Soil erosion can threaten water, soil and air resources; thus, potential soil erodibility is used by the USDA as the basis for eligibility in some conservation programs. Currently, the USDA assesses soil erodibility by using the wind erosion equation that was developed in the 1960s (USDA, 2005). Soil erodibility will be assessed in the near future using new technology; this new technology, or Wind Erosion Prediction System (WEPS), has the capability of simulating potential soil and PM10 loss from agricultural fields. Quantitative data regarding soil and PM10 loss to validate the model and to aid in assessing air quality in communities downwind of eroding fields, however, are unavailable from the Columbia Plateau region of the Pacific Northwest. Therefore, the aim of this study was to assess soil and PM10 loss from conventional winter wheat–summer fallow fields within the Columbia Plateau region of the Pacific Northwest.

Materials and Methods

Soil erosion and loss of PM10 were assessed from agricultural fields located within the Columbia Plateau of eastern Washington (Figure 1). The Columbia Plateau is confined by the Okanogan Highlands to the north, Bitterroot Mountains to the east, Blue Mountains to the south and Cascade Mountains to the west. Sagebrush-steppe vegetation predominates in the western part of the plateau, where annual precipitation is <300 mm, while meadow-steppe vegetation predominates in the east, where annual precipitation is >500 mm. Winds are predominately from the south and west and are typically the strongest in spring and autumn. Wind erosion occurs throughout the plateau, but is particularly acute in areas with <300 mm of annual precipitation and where winter wheat–summer fallow is the predominate dryland cropping system. This cropping system is typified by a 14 month fallow period that begins after harvest in July.

Loss of soil and PM10 was assessed from a 300 m × 300 m field in 2003 and a 400 m × 400 m field in 2004. The field sites were located near Washtucna, Adams County, WA (46°50′N, 118°30′W, elevation of 510 m), but separated by a distance of 10 km. Both field sites were dominated by Ritzville silt loam (Andic Aridic Haplustoll) on less than a 2 per cent slope. Ritzville silt loam is derived from loess more than 1 m thick and occupies over 20 per cent of the landscape in Adams County. A nonerodible surface consisting of winter wheat or wheat stubble in 2003 and permanent grass in 2004 was maintained upwind (south and west) of the sites. The surface downwind (north and east) of the field sites was maintained in winter wheat, wheat stubble, or permanent grass in 2003 and in summer fallow in 2004. Each field site was managed by the cooperator (wheat grower) using conventional tillage practices to control weeds and conserve soil water during the fallow cycle. The site used in 2003 was in wheat production in 2001 and subsequently maintained in conventional fallow in 2002 and 2003. In 2003, the site was cultivated to a depth of 0·1 m with a double-disk implement on 10 April and then rod-weeded on 6 May, 18 June, 4 September, 18 September and 21 October. The field site used in 2004 was in wheat production in 2003 and subsequently cultivated to a depth of 0·10 m with 0·4 m sweeps on 20 October 2003 and a double-disk implement on 20 April 2004 and then rod-weeded on 10 May and 23 June prior to sowing winter wheat on 27 August.
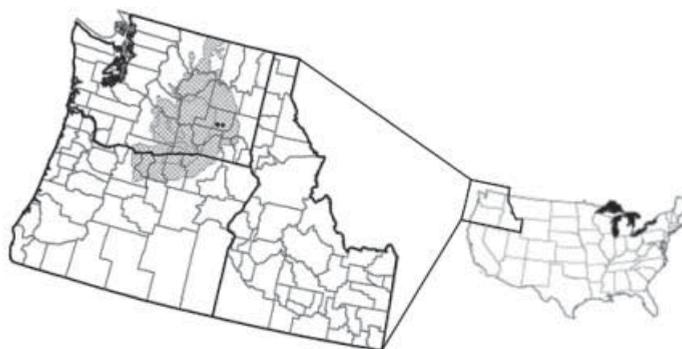

Figure 1. Location of field sites (dots) within the Columbia Plateau (shaded area) of eastern Washington and northern Oregon.
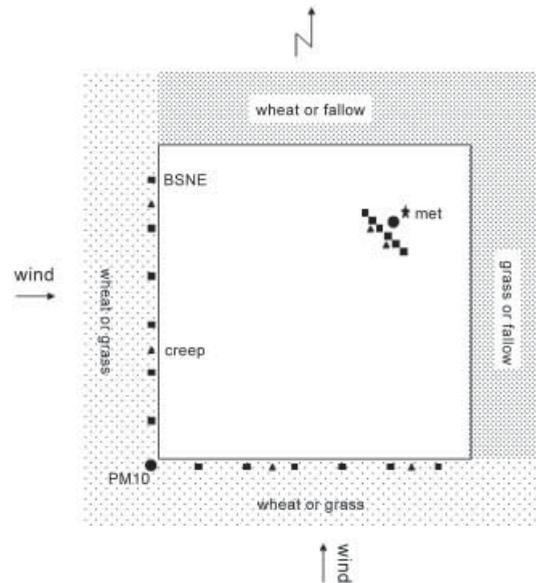
Figure 2. Instrumentation at the field site with symbols representing placement of PM10 samplers, BSNE and creep collectors, and meteorological (met) station.

The field sites were instrumented to monitor horizontal soil flux and PM10 concentrations at the windward (south and west boundary) and leeward (northeast corner) positions in the field (Figure 2). Instruments located at the leeward position in the field were placed 75 m from the northeast corner in 2003 and 100 m from the northeast corner in 2004, thus providing a minimum fetch of 225 m in 2003 and 300 m in 2004. High-volume PM10 samplers (model PM 10, Graseby-Andersen, Village of Cleves, OH) and creep and BSNE collectors (Custom Products and Consulting, Big Spring, TX) were deployed at the windward and leeward positions in the field to respectively assess the influx and efflux of PM10 and soil. One high-volume PM10 sampler was mounted at heights of 3 and 5 m above the soil surface in 2003 and at 1·5, 3, and 6 m above the surface in 2004. The samplers were activated (using 120 V AC electrical generators) when the wind speed exceeded 6·4 m s^{-1} at a height of 3 m for 10 consecutive minutes. This threshold, designated as a high-wind event, is that required to initiate movement of soils across the Columbia Plateau (Saxton *et al*., 2000). The samplers were deactivated when wind speed was less than 5·8 m s^{-1} for 15 minutes. Filters for the high-volume PM10 samplers were equilibrated to standard laboratory conditions prior to weighing before and after deployment. Time-integrated PM10 concentrations obtained using high-volume samplers were augmented with near real-time PM10 concentrations measured at the downwind field position with tapered element oscillating microbalances (model 1400, Rupprecht and Patashnick, Albany, NY) in 2003 and E-samplers (Met One Instruments, Grants Pass, OR) in 2004. Near real-time PM10 concentrations (sampled at 0·5 and 1 Hz and averaged every 10 and 15 minutes for the microbalances and E-sampler, respectively) aided in screening data to ensure that winds were predominately from the south or west (between 160 and 290 degrees) during periods of active PM10 emissions. High-wind event data were discarded for use in this study when winds were from <160 or >290 degrees during the event.

Six sets of BSNE collectors were deployed at the windward and leeward positions in the field to measure saltation and suspension. One set of collectors consisted of five BSNE collectors mounted on a pole at heights of 0·1, 0·2, 0·5, 1, and 1·5 m. Two creep collectors were deployed at each field position to measure discharge to a height of 0·025 m. Sample collections were periodic due to the remoteness of the field sites and generally occurred immediately after a high-wind event. Sediment collected by BSNE and creep samplers was air-dried prior to weighing. For those events with sufficient sediment catch in the BSNE (more than 0·5 g), the sediment was separated into 10, 45, 100, and 150 μm diameter size fractions using a sonic sieve (Gilson, Worthington, OH). Since the BSNE is inefficient in collecting all suspended sediment (Goossens and Offer, 2000), we ascertained the catch efficiency of the BSNE for suspended Ritzville silt loam sediment (particle size <125 μm) and PM10. Catch efficiency was determined in a wind tunnel by (1) placing a 50 mm extension on the front of a BSNE collector, (2) attaching a funnel to the top of the extension and (3) introducing a known amount of sediment or PM10 into the collector via the funnel. Catch efficiency was determined at wind speeds (measured using a pitot tube located adjacent to the opening of the BSNE collector) of

5, 10 and 18 m s^{-1} and computed as the ratio of mass of sediment or PM10 collected in the BSNE to the amount of sediment or PM10 introduced into the collector.

An automated micrometeorological station was deployed at the leeward position in the field to continuously measure wind speed and direction, precipitation, solar radiation, atmospheric temperature and relative humidity. Wind speed (model 14A three-cup anemometer, Met One, Grants Pass, OR) and air temperature (fine-wire thermocouples) were measured at heights of 0·1, 0·5, 1, 2, 3 and 5 m. Micrometeorological sensors were monitored every 10 s and data recorded every 30 minutes except during high-wind events, when data were recorded at 10 minute intervals.

Vertical PM10 flux, F_v, was estimated for each high-wind event according to

$$F_v = \frac{ku_*(C_1 - C_2)}{\ln(z_2/z_1)} \tag{1}$$

where k is von Karman's constant (0·4), u_* is friction velocity (m s^{-1}) and C_1 and C_2 are PM10 concentrations (μg m^{-3}) at heights z_1 and z_2, respectively. Friction velocity was obtained from the logarithmic wind profile equation:

$$u_z = \frac{u_*}{k} \ln(z/z_o) \tag{2}$$

where u_z is the wind speed (m s^{-1}) at height z (m) and z_0 is roughness height (m). Both u_* and z_0 were determined by regression analysis of u_z versus z.

Total horizontal soil flux at the windward and leeward positions in the field was equivalent to the sum of creep and BSNE sediment catch. The vertical distribution of saltating and suspended sediment captured by the BSNE collectors was described using the following equation:

$$q = az^{-b} \tag{3}$$

where q is sediment catch (kg m^{-2}), z is height (m) of the opening of the BSNE collector above the soil surface and a and b are fitted parameters (Zobeck and Fryrear, 1986). Saltating and suspended sediment flux was then determined by integrating (3) from 0·025 to 5 m (approximate height where integrated flux for all high-wind events approached maximum value). Net soil loss from the field for each high-wind event was calculated as the difference between total horizontal soil flux at the leeward and windward positions in the field. This methodology of computing soil loss necessitated using data only for those events characterized by southerly or westerly winds.

Near soil-surface and crop residue characteristics (soil water content and potential, crust cover and thickness, random roughness, bulk density, aggregate size distribution, aggregate stability, and residue biomass and cover) were assessed at three locations within the field before each high-wind event or after each precipitation and tillage event. While these characteristics are important for interpreting differences in soil and PM10 loss among dates, we defer discussion of these characteristics (except aggregate size distribution) to a companion paper, which addresses WEPS model validation using the acquired field data (Feng and Sharratt, submitted). Aggregate size distribution of parent soil was determined using a rotary sieve (Lyles *et al*., 1970) and sonic sieve apparatus with screen sizes of 19, 6·4, 2·0, 0·84, 0·42, 0·150, 0·100, 0·045, and 0·010 mm.

Results and Discussion

Loss of soil and PM10 from a field in conventional summer fallow was assessed for four high-wind events in 2003 and two events in 2004 (Table I). The most severe of these events occurred from 27 to 29 October 2003, when high winds resulted in the greatest horizontal soil flux and PM10 concentrations over the two years of this study. Elevated PM10 concentrations as a result of wind erosion occurred on 28 October 2003 when winds at a height of 3 m exceeded 6·4 m s^{-1} for 14 consecutive hours (Figure 3). The highest 10 minute average PM10 concentration, 8535 μg m^{-3} as measured by a tapered element oscillating microbalance at a height of 5 m, was observed at 1600 h and preceded peak 10 minute average 3 m wind velocities of 17·6 m s^{-1} at 2000 h. The development of a thin and patchy crust at the soil surface, as visually observed by the authors at the field site, in response to precipitation beginning at 1700 h may have suppressed the emission and thus concentration of PM10 during the time of maximum wind velocities. Winds were predominately from the WSW. The magnitude of this event was recorded by the Washington Department of Ecology at Kennewick, WA, located 90 km southwest of the field site. The Washington Department of Ecology (2003) reported a mean daily PM10 concentration of 1438 μg m^{-3} on 28 October 2003, the second highest concentration

Table I. Wind characteristics, horizontal soil flux and PM10 concentrations observed during high-wind events at a field site near Washtucna, WA, in 2003 and 2004

Year	Day	Duration[1] (h)	Wind[2] mean (m s^{-1})	Wind[2] maximum (m s^{-1})	Soil flux[3] (kg m^{-1})	PM10 concentration[4] mean (μg m^{-3})	PM10 concentration[4] maximum (μg m^{-3})
2003	12–22 Sep	26	6·7	12·5	0·46	59	625
	3–15 Oct	47	7·6	14·4	1·43	87	1705
	15–27 Oct	41	7·2	11·9	0·75	22	425
	27–29 Oct	14	10·3	17·6	22·47	791	8535
2004	6–23 Aug	20	6·9	10·3	1·63	152	3650
	23 Aug–9 Sep	43	7·1	10·7	15·12	454	4985

[1] Number of hours that wind speed was in excess of 6·4 m s^{-1} at a height of 3 m.

[2] Mean and maximum wind speed at 3 m height as determined from 10 minute average wind speeds during high-wind events.

[3] Horizontal soil flux integrated over a height of 0–1·5 m.

[4] PM10 concentration at the leeward position in the field measured at a height of 5 m in 2003 and 6 m in 2004; mean concentration was measured by a high-volume sampler whereas maximum concentration was measured by a tapered element oscillating microbalance in 2003 and an E-sampler in 2004.
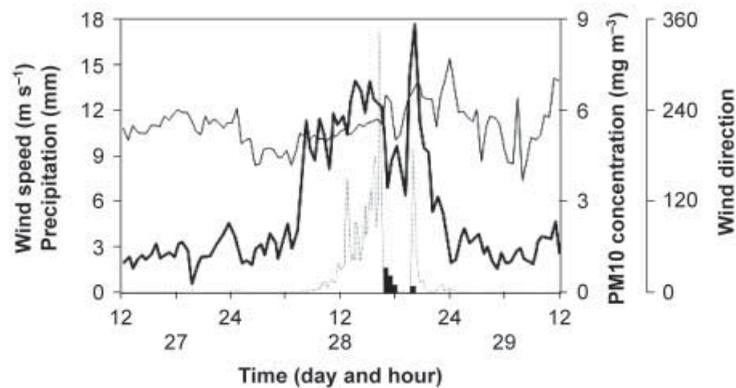
Figure 3. Wind speed (thick line), wind direction (thin line), PM10 concentration (dotted line) and precipitation (bars) at the field site during the 27–29 October 2003 high-wind event.

observed in Kennewick since recording began in 1987 and well above the US EPA National Ambient Air Quality Standard of 150 μg m^{-3} d^{-1}.

Horizontal soil flux diminished with height above the soil surface for each high-wind event. The soil flux profile is illustrated in Figure 4 for two events that resulted in the greatest soil loss over the two years of this study, namely for the 27–29 October 2003 and 23 August–9 September 2004 events. Little or no soil was captured by the BSNE collectors at the windward position in the field during each event, thus indicating that the non-erodible boundaries were effective in minimizing soil transport into the field when winds were predominately from the south and west.

Soil loss, or the amount eroded from the field, was determined by subtracting the horizontal flux (from the soil surface to a height of 5 m) at the windward position from that at the leeward position in the field. Since horizontal soil flux is dependent upon the sediment catch efficiency of the BSNE collector, the catch efficiency of the collector was determined for suspended Ritzville sediment in a wind tunnel. The sediment catch efficiency of the collector ranged from 55 to 65 per cent over a range in wind tunnel free stream velocities of 5–18 m s^{-1}. For the purpose of this study, we assumed that the catch efficiency of the BSNE collector for suspended sediment was 60 per cent. Soil loss, based upon this catch efficiency, ranged from 4 g m^{-2} or 40 kg ha^{-1} for the 12–22 September 2003 high-wind event to 232 g m^{-2} or 2320 kg ha^{-1} for the 27–29 October 2003 event (Table II). The highest loss observed in this study (equivalent to 2·3 Mg ha^{-1}) was comparable to that measured from a silt loam during a single erosion event in Colorado and Washington. Indeed, Van Donk and Skidmore (2003) reported a loss of 0·6 Mg ha^{-1} from a 135 ha field in eastern Colorado while Zobeck *et al.* (2001) reported a loss of 1·5 Mg ha^{-1} from a 3 ha field in south central Washington. Loss observed in this study, however, was 25 times smaller than the loss (56 Mg ha^{-1}) measured from a
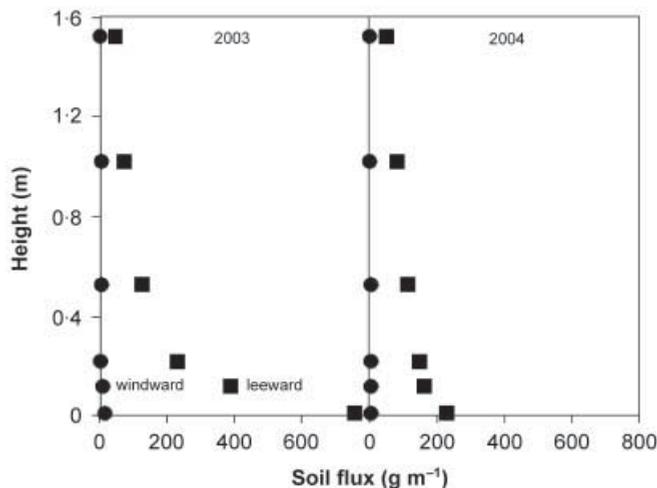
Figure 4. Horizontal soil flux as a function of height above the soil surface at the windward and leeward field positions for the 27–29 October 2003 and 23 August–9 September 2004 high-wind events.

Table II. Height of PM10 plume and loss of soil and PM10 from a fallow field near Washtucna, WA, due to high-wind events in 2003 and 2004

Year	Day	Plume height (m)	Soil loss[1]		PM10 loss[2]	
			actual (g m^{-2})	potential (g m^{-2})	(g m^{-2})	(%)
2003	12–22 Sep	6	2·5	4·3	0·5	11·4
	3–15 Oct	5	5·1	11·8	1·0	8·6
	15–27 Oct	6	2·6	4·4	0·5	11·6
	27–29 Oct	6	102·1	231·7	21·2	9·1
2004	6–23 Aug	7	4·9	13·8	1·6	11·6
	23 Aug–9 Sep	7	50·6	160·4	16·3	10·2

[1] Actual soil loss determined from creep and BSNE sediment catch within 1·5 m of the soil surface. Potential loss accounts for sampling inefficiency of the BSNE collector and suspended sediment to 5 m height.

[2] PM10 loss between 0·025 m and height of plume as determined by BSNE collectors and high-volume PM10 samplers. Percent loss relative to potential soil loss.

3 ha field of loamy sand during a single event in Texas (Zobeck *et al.*, 2001), about 15 times smaller than the loss (30 Mg ha^{-1}) observed from a 3 ha field of clay loam during a single event in Alberta (Larney *et al.*, 1995) and five times smaller than the loss (12 Mg ha^{-1}) observed from a 3 ha field of loamy sand during a single event in Washington (Zobeck *et al.*, 2001). Differences in soil loss among studies may be partially attributed to dissimilarities in soil texture, but other factors such as duration of event, wind speed, surface roughness and vegetative cover may also contribute to differences in soil loss among studies.

Vertical PM10 flux, computed from Equation (1) using data collected by the high-volume samplers, for the six high-wind events, ranged from 10 µg m^{-2} s^{-1} for the 3–15 October 2003 event to 255 µg m^{-2} s^{-1} for the 27–29 October 2003 event. Four of the six high-wind events observed in this study were characterized by PM10 fluxes in excess of those observed by Kjelgaard *et al.* (2004b) over an agricultural field within the Columbia Plateau; their range in vertical flux observed across four high-wind events did not exceed 25 µg m^{-2} s^{-1}. The highest vertical flux observed in this study, however, is similar to the PM10 flux of 235 µg m^{-2} s^{-1} observed by Gillette *et al.* (1997) at Owens Lake, CA, as well as PM20 (particulates of <20 µm in diameter) fluxes of >300 µg m^{-2} s^{-1} observed by Sabre *et al.* (1997) and Gomes *et al.* (2003) over cultivated agricultural fields in Niger and Spain. Friction velocities estimated using Equation (2) varied from 0·29 m s^{-1} for the 27–29 October 2003 event to 0·67 m s^{-1} for the 6–23 August 2004 event while roughness height ranged from 0·004 m for the 3–15 October 2003 event to less than 0·001 m for the 23 August–9 September 2004 event. Similar u_* and z_0 values were observed by Kjelgaard *et al.* (2004b) within the Columbia Plateau.
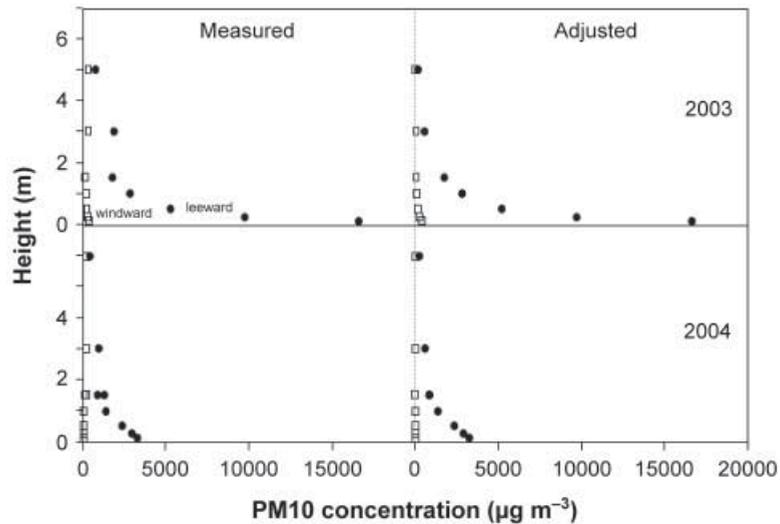
Figure 5. PM10 concentration profile at the windward and leeward field positions for the 27–29 October 2003 and 23 August–9 September 2004 high-wind events. Profiles are shown using the measured and adjusted high-volume sampler concentrations.

PM10 concentration profiles during each high-wind event were ascertained using both BSNE sediment catch and high-volume sampler concentrations. PM10 concentration was determined from BSNE sediment catch according to the equation

$$C_z = \frac{M_z}{u_z Stf} \tag{4}$$

where C_z is PM10 concentration at height z, M_z is the PM10 mass in the BSNE collector at height z, u_z is wind velocity at height z, S is the area of BSNE opening, t is the duration of an event and f is the PM10 catch efficiency of BSNE collectors. The PM10 catch efficiency of the BSNE collector for Ritzville silt loam, as determined by mass balance in a wind tunnel, ranged from 10 per cent at a wind velocity of 18 m s^{-1} to 25 per cent at a wind velocity of 5 m s^{-1}. For the purpose of this study, we assumed that catch efficiency varied with wind speed. Shao *et al*. (1993) estimated the BSNE entrapment efficiency was 40 per cent for particles smaller than 10 μm whereas Goossens and Offer (2000) report efficiencies of 40 per cent for extremely fine soils at wind velocities of 1–5 m s^{-1}. While our efficiencies are smaller than those previously reported, differences in wind speed among studies may affect the efficiency of BSNE collectors.

PM10 concentration profiles appeared to mimic those of horizontal soil flux profiles, but a discontinuity in the PM10 concentration profile at both the windward and leeward positions in the field was apparent at a height of about 1·5 m (between the highest BNSE collector and lowest high-volume sampler) during all high-wind events. This discontinuity in the profile using measured PM10 concentrations is illustrated for the 27–29 October 2003 and 23 August–9 September high-wind events in Figure 5, and suggested a difference in instrument performance with the BSNE collector under-sampling PM10 or the high-volume sampler over-sampling PM10. Differences in instrument performance could be substantiated in 2004 when the instruments were collocated at a height of 1·5 m. During the 23 August–9 September 2004 high-wind event, the PM10 concentration at a height of 1·5 m at the leeward position in the field was 1315 μg m^{-3} as measured by the high-volume sampler and 910 μg m^{-3} based upon BSNE sediment catch. Likewise, during the 6–23 August 2004 high-wind event the PM10 concentration measured by the high-volume sampler was 395 μg m^{-3} while that estimated using BSNE collectors was 210 μg m^{-3}. These discrepancies may be expected due to differences in the design characteristics of the instruments. For example, performance of the instruments will differ due to the passive-sampling, near-isokinetic design of the BSNE collector and the active-sampling, non-isokinetic design of the high-volume sampler. The high-volume sampler appeared to over-sample PM10 as the PM10 sampling efficiency of the BSNE was determined at various wind speeds in this study. Buser *et al*. (2003) also indicated that the performance of the high-volume PM10 sampler is affected by the source of PM10. They found that the high-volume sampler overestimates PM10 concentrations by more than 200 per cent when sources, such as

agricultural soils, are characterized by a mass mean diameter larger than 10 µm. Based upon our findings (calibration of the BSNE collector for PM10 in the wind tunnel and comparison of the collocated BSNE collector and high-volume sampler in the field) and those of Buser *et al*., the high-volume sampler PM10 concentrations were reduced by a factor of 1·5 for all high-wind events to minimize the discontinuity in the PM10 profile (Figure 5) and to improve estimates of PM10 loss. Adjusting the high-volume sampler concentrations by a factor of 1·5 did not affect PM10 loss, as PM10 loss differed by <5 per cent when using the measured or adjusted high-volume PM10 concentrations.

Loss of PM10 from the field site was assessed for each high-wind event by subtracting horizontal PM10 flux at the windward position from that at the leeward position in the field. Horizontal PM10 flux at each field position was determined by

$$PM10 \; flux = \int C_z u_z t dz \tag{5}$$

where PM10 flux is in $\mu g \; m^{-1}$. The integral was evaluated from 0·025 m to the height of the PM10 plume. Plume height was obtained by extrapolating the PM10 concentration profiles (3 and 5 m heights in 2003 and 1·5, 3 and 6 m heights in 2004) at the windward and leeward positions in the field; plume height was the height where the tops of the profiles intercepted. For example, plume height for the 27–29 October 2003 event was 6 m whereas plume height for the 23 August–9 September 2004 event was 7 m based upon the extrapolated profiles (Figure 5). Plume height for the 27–29 October 2003 event closely matched visual observations made at the field site on 28 October 2003. Loss of PM10 between the windward and leeward positions in the field ranged from 0·5 g m^{-2} or 5 kg ha^{-1} for the 12–22 September 2003 and 15–27 October 2003 events to 21 g m^{-2} or 210 kg ha^{-1} for the 27–29 October 2003 event (Table II). Assuming that near-surface bulk density is 1 Mg m^{-3} (Feng and Sharratt, submitted), loss of PM10 for the 27–29 October 2003 event would constitute 2·1 per cent of the mass in the uppermost 1 mm of the soil profile. This percentage was greater than that found in the parent soil before the high-wind event; the parent soil contained 0·9 per cent PM10. The discrepancy between PM10 content of the parent soil and PM10 loss is likely a result of breakage or abrasion of aggregates or incorrectly specifying the depth to which the soil is influenced by wind and saltating particles. No comparable information regarding PM10 loss was found in the literature.

Loss of PM10 from agricultural soils influences air quality within the Columbia Plateau. The significance of PM10 loss observed in this study can be illustrated by assuming a field with dimensions of 100 m × 100 m, PM10 emissions occur from the field on a day when winds of 8 m s^{-1} are sustained for 10 hours (Stetler and Saxton, 1996), PM10 is emitted into an affected area (cross-sectional length of 100 m) immediately downwind of the field and uniformly mixes to a height of 400 m within the atmospheric boundary layer (Claiborn *et al*., 1998) and winds are calm prior to and following the high-wind event. Based upon these field and atmospheric characteristics, emission of 5 kg PM10 during the high-wind event will effectively increase atmospheric PM10 concentration by 0·43 µg m^{-3} within the affected area. Thus, the atmospheric PM10 concentration would exceed the US EPA National Ambient Air Quality Standard of 150 µg m^{-3} when more than 348 fields (>348 ha) immediately upwind of the affected area are each emitting 5 kg PM10 during the event. Similarly, the National Ambient Air Quality Standard would be exceeded when more than nine fields (>9 ha) immediately upwind of the affected area are each emitting 200 kg PM10 during the event. Furthermore, assuming that the cross sectional length of the affected area is representative of regional storms at about 200 km (Claiborn *et al*., 1998), PM10 concentrations within the affected area would exceed the National Ambient Air Quality Standard when >0·7 million ha immediately upwind of the affected area is emitting 5 kg PM10 ha^{-1} or when >17 300 ha upwind of the affected area is emitting 200 kg PM10 ha^{-1}. Although nearly 0·7 million ha are maintained in summer fallow in any given year, the latter case would represent about 2 per cent of all fields in summer fallow within the Columbia Plateau.

Aggregate size analyses on non-dispersed soil samples taken before each high-wind event indicated that about 50 per cent of the parent soil in 2003 and 15 per cent of the parent soil in 2004 was comprised of erodible material (particles with diameters <840 µm). The difference in the erodible fraction between years (Table III) is likely due in part to differences in management practices imposed upon the fields. The field site in 2003 was in the second year of fallow and thus had been subject to multiple tillage and rod weeding operations in 2002 and 2003. In comparison, the field site in 2004 received half as many tillage and rod weeding operations. Nevertheless, a significant fraction of the erodible material was suspension-size particulate. For example, 10 per cent of the erodible fraction was creep size (particulates with diameters of 840–420 µm), 40 per cent was saltation size (420–100 µm) and 50 per cent was suspension size (<100 µm) for the 27–29 October 2003 event (Table III). Likewise, 75 per cent of the erodible fraction was creep size, 10 per cent was saltation size and 15 per cent was suspension size for the 23 August–9 September 2004 event. Although the erodible material on the soil surface was comprised of saltation-size particulate, only a small fraction of the eroded sediment captured by the BSNE collectors was saltation-size particulate. This is illustrated for

Table III. Fraction of parent material and eroded soil that is immobile (>840 μm), creep size (840–420 μm), saltation size (100–420 μm) and suspension size (<100 μm) determined for the 27–29 October 2003 and 23 August–9 September 2004 high-wind events.

Event	Immobile		Creep		Saltation		Suspension	
	Parent[1]	Eroded[2]	Parent	Eroded	Parent	Eroded	Parent	Eroded
2003	0·51	0	0·06	0·03	0·18	0·07	0·25	0·90
2004	0·87	0	0·10	0·02	0·01	0·01	0·02	0·97

[1] Parent material collected from the upper 30 mm of the soil profile.
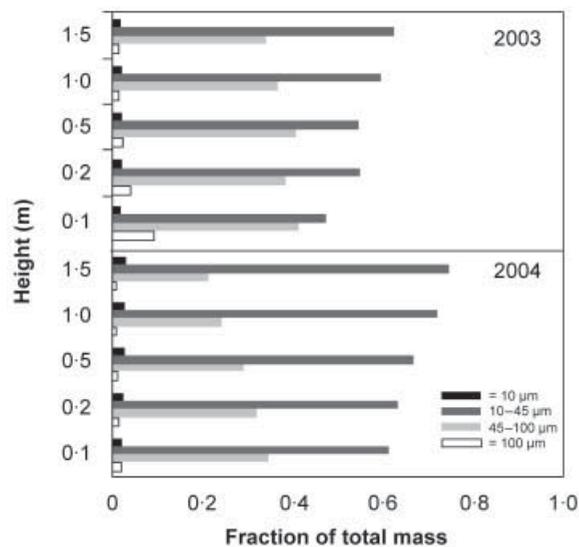[2] Eroded soil as captured by creep and BSNE collectors.

Figure 6. Particle size mass fraction in BSNE collectors positioned at various heights above the soil surface for the 27–29 October 2003 and 23 August–9 September 2004 high-wind events.

the 27–29 October 2003 and 23 August–9 September 2004 events when respectively 7 and 1 per cent of the eroded sediment captured by the BSNE collectors was saltation-size particulate (Table III). The small fraction of saltation-sized particles captured by the BSNE collectors suggests that surface random roughness (10–11 mm) or residue cover (0·02–0·04 m² m⁻²) was sufficient to trap saltating particles or that saltating particles were sufficiently fragile so as to break upon impacting other soil particulates or the collector. Kjelgaard *et al.* (2004b) reported little or no saltation activity for a Ritzville silt loam during high-wind events on the Columbia Plateau and suggested that direct suspension is the dominant process by which soils erode. Our results provide evidence to support their conclusion.

Aggregate size analyses of the eroded sediment captured by the BSNE collectors suggested that the proportion of particles smaller than 45 μm increased with height. Similar trends are found in data presented by Goossens (1985) and Kjelgaard *et al.* (2004a), but only Goossens (1985) reported a noticeable shift in the particle size mode with height. An apparent shift in particle size mode with height was found in this study as illustrated for the 27–29 October 2003 and 23 August–9 September 2004 events in Figure 6. For example, the sediment fraction larger than 100 μm ranged from 9 per cent at a height of 0·1 m to 1 per cent at a height of 1·5 m, whereas particles smaller than 45 μm accounted for 49 per cent of the sediment catch at a height of 0·1 m and 64 per cent of the catch at a height of 1·5 m for the 27–29 October 2003 event. Particles smaller than 10 μm accounted for about 2 per cent of the BSNE sediment catch at heights of 0·1 and 1·5 m. This small fraction may be attributed in part to the inefficiencies of the BSNE collectors in trapping small particles (Goosens and Offer, 2000).

Conclusions

Soil and PM10 loss from a silt loam during the fallow cycle of a winter wheat–summer fallow rotation ranged from 40 to 2320 kg ha^{-1} and from 5 to 210 kg ha^{-1}, respectively, across six high-wind events observed in 2003 and 2004 within the Columbia Plateau. Loss of PM10 constituted 9–12 per cent of the total soil loss observed during the events. Loss of soil as a result of high winds appeared to be driven by direct suspension rather than saltation processes. Thus, land management practices (e.g. minimum tillage to retain surface residue) or crop rotations (e.g. annual) should be developed or identified that will minimize emission of fine soil particles that are most vulnerable to direct suspension.

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