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Wind erosion potential of a winter wheat–summer fallow rotation after land application of biosolids

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ABSTRACT

Conservation tillage is a viable management strategy to control soil wind erosion, but other strategies such as land application of biosolids that enhance soil quality may also reduce wind erosion. No studies have determined the effects of biosolids on wind erosion. Wind erosion potential of a silt loam was assessed using a portable wind tunnel after applying synthetic and biosolids fertilizer to traditional (disk) and conservation (undercutter) tillage practices during the summer fallow phase of a winter wheat-summer fallow (WW-SF) rotation in 2015 and 2016 in east-central Washington. Soil loss ranged from 12 to 61% lower for undercutter than disk tillage, possibly due to retention of more biomass on the soil surface of the undercutter versus disk tillage treatment. In contrast, soil loss was similar to or lower for biosolids as compared with synthetic fertilizer treatment. Our results suggest that biosolids applications to agricultural lands will have minimal impact on wind erosion.

1. Introduction

Sustainable agriculture involves the production of food, fiber, and/or feed with minimal harm to ecosystems, animals or humans (Seufert et al., 2012). For decades, biosolids generated from wastewater treatment plants have been applied to agricultural fields to replace fertilizer as a sustainable management strategy in the United States (Bhat et al., 2013). With an ever growing world population, the generation of wastewater and biosolids will continue to rise.

Wastewater or sewage sludge can be polluted as a result of harboring enteric bacteria, pathogenic organisms, heavy metals, and particulate matter (Bhat et al., 2013). Since the 1950s, federal legislation in the United States has been strengthened to control water pollution (Lu et al., 2012). To reduce pollution, wastewater undergoes physical, chemical, and biological treatment at treatment facilities. Effluent resulting from primary treatment of wastewater can be further processed for discharging into surface water systems or the irrigation of crop or public land. Sewage sludge, or the semi-solid slurry collected during primary treatment of wastewater, contain organisms, chemicals, or particulate matter in concentrations that are harmful to humans (Lu et al., 2012). For this reason, sewage sludge undergoes digestion to stabilize organic matter and reduce levels of harmful organisms. The solids that remain after digestion are then referred to as biosolids which are typically dewatered before being transported for application on fields.

Applying biosolids to agricultural land in lieu of synthetic fertilizers is a relatively safe method to recycle or sustainably use biosolids even though biosolids originate from sewage sludge (Lagae et al., 2009). In addition, there is an economic benefit to using biosolids in agroecosystems (Lu et al., 2012). Concerns about the economic, environmental, and social impacts of traditional agricultural practices have led many farmers to seek more sustainable practices (Reganold et al., 1993). With proper application techniques (e.g. rate, depth), biosolids may be a more economical and sustainable practice than use of synthetic fertilizer to meet the nutrient requirements of crops (Lagae et al., 2009; Cogger et al., 2013). Biosolids also contain organic matter which is vital to maintaining soil health (Brady, 1990).

Biosolids are typically applied to agricultural lands in arid and semiarid regions because of the low risk for runoff after precipitation events (Cogger et al., 2001). Wind erosion, however, can be an environmental concern in arid and semiarid regions due to emission of particulates into the atmosphere. Wind erosion could thus potentially transport surface-applied biosolids offsite (Lagae et al., 2009). Liquid or dewatered biosolids are typically spread onto the soil surface and then incorporated into the soil to minimize odors and volatilization of ammonia (Lu et al., 2012). Biosolids may influence soil erodibility as they

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contain relatively high amounts of organic matter and nutrients. Altering the chemical and/or biological composition of the soil may influence soil physical properties and/or plant growth and thus the erodibility of the soil. For example, Garcia-Orenes et al. (2005) found an increase in aggregate stability after applying dewatered biosolids to degraded soils. Wallace et al. (2010) also found an increase in aggregate size and stability after applying dewatered biosolids to a semiarid soil in Canada. Soil particle or aggregate size has a dramatic effect on wind erosion processes as more energy is required to lift and transport larger particles or aggregates (Zobeck, 1991). The increase in aggregate size noted by Wallace et al. (2010) may reduce wind erosion from agricultural land based on the reduction in the soil erodibility factor (Pi et al., 2017).

Previous studies related to wind erosion or windblown particulate emissions from soil amended with biosolids have focused on transport of pollutants during application. Paez-Rubio et al. (2006) found that PM10 (particulate matter ≤10 μm in diameter) emissions were three times lower when disking agricultural soils that had been amended versus non-amended with surface-applied dewatered biosolids. They attributed the reduction in PM10 emissions to higher moisture content of the amended versus non-amended soil. Similarly, Akbar-Khanzadehet al. (2012) reported that application of dewatered biosolids to dry soil resulted in an increase in soil moisture and subsequent reduction in particulate emissions when incorporating biosolids into the soil. Bhat and Kumar (2012) found particulate emissions highest during, rather than before or after, application of liquid biosolids to agricultural soils in Ohio. They suggested that disturbing the soil during injection of biosolids (to a depth of 0.6 m) was one possible reason for the high particulate emissions. Paez-Rubio et al. (2007) measured particulate emissions as dewatered biosolids were applied to agricultural soils using a manure spreader in Arizona. They found 7.6 mg of biosolids were aerosolized for every 1 kg of biosolids that were applied to soils.

Soil amended with biosolids could be especially valuable in the WW-SF region of the Inland Pacific Northwest (PNW) where conservation-tillage and no-tillage management practices have failed to significantly bolster soil organic matter due to relatively low stable carbon content of wheat residue and modest production levels (Gollany et al., 2013). There is, however, potential risk associated with biosolid particulates being emitted from the soil surface into the atmosphere and adversely affecting air quality. Sullivan et al. (2007) suggested that enhancements in soil physical properties (e.g. aggregation, water retention, infiltration) resulting from applications of biosolids can reduce soil erosion, thus minimizing the risk of off-site transport of biosolid particulates.

Wind erosion is of paramount importance in the low-precipitation zone (< 300 mm of annual precipitation) of the PNW where 1.5 million ha are managed in a WW-SF rotation (Schillinger et al., 2006). Emission of soil particulates from land managed in conventional tillage-based WW-SF has caused road closures and exceedance of National Ambient Air Quality Standards for PM10 during high wind events (Sharratt and Lauer, 2006). For this reason, Sharratt and Feng (2009) and Sharratt et al. (2010) focused attention on conservation or reduced tillage practices in WW-SF to control wind erosion in the region. While undercutter tillage has been proven to reduce wind erosion by up to 65% as compared with conventional tillage in WW-SF (Sharratt and Feng, 2009), other methods are yet sought to control wind erosion. We are not aware of published reports that document the influence of biosolids application on wind erosion of agricultural lands which, if successful, will improve air quality in the region. Thus, the focus of this study was to determine the wind erosion potential associated with application of biosolids to land in a WW-SF rotation.

2. Materials and methods

2.1. Field site description

Field experiments were conducted in 2015 and 2016 at the Washington State University Dryland Research Station located near Lind, Washington (47°00′N, 118°34′W). Lind is located in the low precipitation zone of east-central Washington and receives 242 mm of average annual precipitation. Winter wheat-summer fallow is the dominate rotation practiced throughout the region. The experiment was conducted on a Ritzville silt loam (coarse-silty, mixed, superactive, mesic Calcidic Haploxerolls) in 2015 and a Shano silt loam in 2016. Although sets of experimental plots were located 0.7 km apart and subject to the same climatic conditions, Ritzville silt loam had a geometric mean particle diameter of 24 μm, 13% clay, 61% silt, 26% sand, and 0.7% organic matter while Shano silt loam had a geometric mean particle diameter of 31 μm, 9% clay, 56% silt, 35% sand, and 0.7% organic matter. Soils in the WW-SF region of the PNW are highly susceptible to wind erosion due to meager precipitation, low biomass production, and multiple tillage operations during the fallow phase of the rotation. High winds typically occur in March-April and September-October and coincide with primary tillage during the fallow phase of the rotation in spring and the subsequent sowing of winter wheat in late summer (Papendick, 2004).

Our experiment was configured as a split-block design with four replications. Main plot treatments were tillage and subplot treatments were fertilizer. Tillage treatments were traditional tillage using a tandem disk implement and conservation tillage using an undercutter implement. Size of individual main plots were 76 × 8 m and subplots were 38 × 8 m. Fertilizer treatments were either synthetic or biosolids fertilizer.

Wind erosion assessments were made after primary tillage and sowing winter wheat in 2015 and 2016, although on different sets of experimental plots. The plots were harvest the preceding July and were not disturbed until Glyphosate herbicide [N-(phosphonomethyl) glycine] was applied at rate of 0.43 kg acid equivalent ha^{-1} to control weeds in mid-April 2015 and 2016. Class B biosolids, obtained from the King County Wastewater Treatment Division, Seattle, Washington, were then applied using a manure spreader to one set of experimental plots on 4 May 2015 and to the other set of experimental plots on 19 April 2016. Biosolids were applied at a rate to meet the nutrient requirements for two crop years. Thus, experimental plots used in 2015 received their first application of biosolids in 2011 and experimental plots used in 2016 received their first application of biosolids in 2012. The dewatered biosolids were applied at a rate of 6508 kg ha^{-1} to traditional and conservation tillage (hereafter referred to as respectively disk and undercutter tillage) main plots during the fallow phase of the rotation. Synthetic fertilizer (56 kg N ha^{-1} plus 11 kg S ha^{-1}) was broadcast on the surface of plots prior to disk tillage or injected into the soil during undercutter tillage. Immediately (within 2 h) after surface-applying biosolids or synthetic fertilizer (4 May 2015 and 19 April 2016), the plots were tilled to a depth of 0.13 m in both the disk and undercutter tillage treatments (note that synthetic fertilizer was injected below the soil surface with the undercutter during tillage). The undercutter implement has 0.8-m wide sweeps that slice beneath the soil with minimal disturbance of surface crop residue. In 2015, plots were rodweeded to a depth of 0.1 m on 15 June and sown to winter wheat on 8 September using a deep furrow grain drill. In 2016, plots were rodweeded on 3 June and 10 July and sown to winter wheat on 2 September.

2.2. Measurement of wind erosion potential

Horizontal sediment flux representative of a high wind event in the PNW was assessed soon after rodweeding and sowing operations using a portable wind tunnel. Sediment flux of treatments was measured on 18 June and 10 September 2015 and on 6 June and 7 September 2016.

These dates correspond to three days after rodweeding plots both years and two and five days after sowing plots in respectively 2015 and 2016. Soils are most susceptible to wind erosion after tillage in spring and sowing winter wheat in late summer due to high winds, dry soils, and low residue cover (Papendick, 2004). We were unable to assess sediment flux after primary tillage in spring due to the reentry time (30 d) after field application of biosolids. Therefore, we chose to assess sediment flux after the first rodweeding in June. Precipitation did not occur between the time of tillage or seeding and measurement of sediment flux, except after sowing wheat in 2016. Although 0.2 mm of precipitation occurred four days after sowing wheat in 2016 and one day prior to measuring sediment flux, this trace precipitation event appeared to have little impact on soil physical properties as the soil did not crust.

The portable wind tunnel has a working section 7.3 m long, 1.2 m high and 1.0 m wide. The wind tunnel is transported on a trailer and the working section positioned over the soil surface with the aid of a hydraulic crane. A 33 kW fan can generate winds of 20 m s^{-1} inside the tunnel. Airflow is conditioned before finally passing through a grid assembly which initiates shear flow inside the working section of the tunnel. Airflow is conditioned to achieve shear flow characteristics that occur naturally in the field (Pietersma et al., 1996; Sharratt, 2007). Horizontal sediment flux was measured at a wind speed of 16 m s^{-1} for an initial 10 min. This wind speed reoccurs about once every two years in east-central Washington (Wantz and Sinclair, 1981). Immediately (~30 s) thereafter, sediment flux was measured for another 10 min but under active (rather than limited) saltation conditions. Active saltation activity characteristic of field conditions was achieved by introducing an abrader (sand particles 250–500 μm in diameter) into the air stream of the wind tunnel.

Horizontal sediment flux was measured at a distance of 5.4 m downwind of the grid assembly using an isokinetic slot sampler (Stetler et al., 1997) that traps saltating and suspended sediment to a height of 0.75 m above the soil surface. The slot sampler was calibrated in the wind tunnel prior to the experiment to ensure isokinetic conditions at a free stream velocity of 16 m s^{-1} inside the tunnel. Soil loss was calculated as the ratio of horizontal sediment flux to the length of the soil surface upwind of the sampler (distance from the grid assembly to the slot sampler). Wind speed was measured at six heights from 0.05 to 0.6 m above the soil surface using pitot tubes.

2.3. Surface characteristics influencing wind erosion

In addition to measuring horizontal sediment flux, soil physical properties and surface characteristics were assessed to aid in understanding differences or similarities in soil loss among treatments. Characteristics were assessed at three locations adjacent to but outside of the wind tunnel and at the same time as measurement of sediment flux. Soil water content and potential in the upper 5 mm of the profile was determined by respectively gravimetric analysis and a water activity meter (model MP4, Decagon Devices, Pullman, WA). Surface random roughness and residue flat cover were measured using a pin-type profile meter (Allmaras et al., 1966). Stem area index was determined by the height, diameter, and population of standing residue elements within a 0.5 × 0.5 m rectangular frame. Standing and prostrate crop residue biomass were assessed by collecting, drying, and weighing biomass taken within the rectangular frame. Surface penetration resistance was determined using a pocket penetrometer. Aggregate size distribution was determined on 1-kg samples collected from the upper 30 mm of the soil profile. After the samples were air-dried, they were processed through a compact rotary sieve (Chepil, 1962) equipped with sieves having 0.42, 0.84, 2.0, 6.4, and 19.0 mm openings. The size distribution of the < 0.42-mm size fraction and PM10 fraction was determined using a sonic sieve. Soil bulk density was determined by extracting 30 mm deep cores from the soil surface and then drying the sample at 105 °C prior to measuring the soil dry weight.

2.4. Statistical analysis

Soil loss and surface characteristics of tillage and fertilizer treatments measured on each date were analyzed for differences using commercial software (SPSS Statistics 20.0; The SPSS Inc., Chicago, IL). Prior to performing a split-block Analysis of Variance (ANOVA), the data were examined for homogeneity of variance across treatments and normalcy of distribution within treatments. Homogeneity of variance was tested using residual plots and normalcy of distribution was tested using the normal probability plot in SPSS. In the event significant F-values ($P \leq 0.05$) were found using ANOVA, differences between treatments means were separated using Least Significant Difference (LSD). Treatment differences in soil loss across measurement dates were analyzed for differences using Multi-factor Analysis of Variance (MANOVA), nonparametric test, paired-samples *t* test, and regression analysis. A nonparametric test was used when the data exhibited heterogeneous variance or non-normal distribution, which only occurred when comparing tillage or fertilizer treatments across measurement dates.

3. Results and discussion

Soil loss measured from synthetic and biosolids fertilizer treatments that were applied to disk and undercutter tillage is shown in Fig. 1. Soil loss was measured over two consecutive 10 min sample periods after rodweeding and sowing winter wheat in 2015 and 2016, the first period representing limited saltation conditions and the second period representing active saltation conditions. Soil and residue characteristics at the time soil loss was measured from each treatment is shown in Table 1.

3.1. Soil loss from synthetic versus biosolids fertilizer

No statistical differences in soil loss were found between fertilizer treatments except after rodweeding on 06 Jun 2016 according to ANOVA (Tables 2 and 3). Soil loss on this date was lower for the biosolids than the synthetic fertilizer treatment by 31%, but only under active saltation conditions imposed by adding abrader to the airstream. Although no statistical differences in surface characteristics between biosolids and synthetic fertilizer treatments were detected on 06 Jun 2016 (Table 1), subtle differences in multiple surface characteristics may affect soil loss. On this date, for example, aggregate geometric mean diameter and surface water content were respectively 13 and 27% higher while soil water potential was 15% lower for biosolids than synthetic fertilizer (Table 1). Differences in other surface characteristics between fertilizer treatments did not exceed 8%. Larger aggregates and wetter soils have been shown to increase the threshold wind speed, reduce soil surface friction velocity, and enhance trapping of saltating particles (Hagen, 1996). Thus, the seeming greater GMD and higher surface water content for biosolids than synthetic fertilizer treatment on 06 Jun 2016 may offer greater protection to the soil surface against wind erosion. Previous studies (Bhat and Kumar, 2012; Krause, 1988) also suggest that biosolids can influence soil aggregation. Krause (1988) found biosolids enhanced aggregate size and stability due to an increase in soil organic matter. We found soil organic carbon content was higher for the biosolids than synthetic fertilizer treatment (data not shown). Averaged across dates and tillage treatments, organic carbon content was 0.7% for synthetic fertilizer and 1.0% for biosolids fertilizer treatments.

3.2. Soil loss from disk versus undercutter tillage

Soil loss appeared to differ between disk and undercutter tillage treatments (Fig. 1). Differences in soil loss between treatments were significant ($P \leq 0.05$) after rodweeding both years (Tables 2 and 3). For example, soil loss from undercutter tillage was 44 and 57% lower than

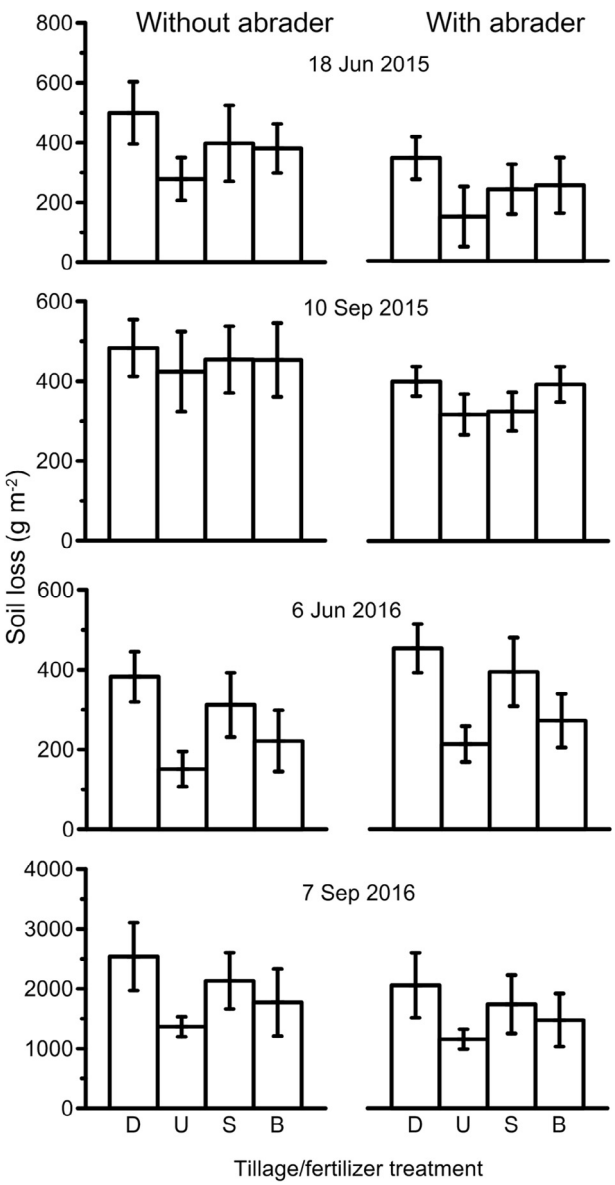

Fig. 1. Soil loss from synthetic (S) and biosolids (B) fertilizer treatments applied to disk (D) and undercutter (U) tillage. Soil loss was measured after rodweeding on 18 Jun 2015 and 06 Jun 2016 and sowing wheat on 10 Sep 2015 and 07 Sep 2016 under limited (without abrader) and active saltation (with abrader) conditions using a portable wind tunnel. The bars indicate standard deviation.

disk tillage under respectively limited and copious saltation activity conditions on 18 Jun 2015 and 61 and 53% lower than disk tillage under respectively limited and copious saltation activity conditions on 06 Jun 2016. Lower soil loss from undercutter tillage may be due to the greater biomass retained on the soil surface from undercutter versus disk tillage (Table 1). On 18 Jun 2015, for example, prostrate and standing residue biomass, residue stem area index, and residue flat cover were respectively 475, 350, 300, and 170% greater for undercutter than disk tillage. In addition, on 06 Jun 2016, prostrate and standing residue biomass, residue stem area index, and residue flat cover were respectively 742, 564, 500, and 212% greater for undercutter than disk tillage. Significant differences in biomass characteristics between tillage treatments persisted until after sowing wheat, although the differences were generally smaller. Similarly, soil loss was seemingly different between tillage treatments after sowing wheat, although not statistically significant. For example, soil loss from undercutter tillage was 12 to 21% lower than disk tillage under respectively limited and copious saltation activity conditions on 10 Sep 2015 and 46 and 44% lower than disk tillage under respectively limited and copious saltation activity conditions on 07 Sep 2016.

Table 1
Surface characteristics of a windblown soil altered by synthetic and biosolid fertilizer applications and disk and undercutter tillage practices during summer fallow. Characteristics were measured on two dates after applying fertilizer and tillage treatments to experimental plots on 4 May 2015 and 19 April 2016.

Surface characteristic	Date of measurement	Tillage treatment		Fertilizer treatment	
		Disk	Undercutter	Synthetic	Biosolid
Prostrate residue biomass, $g\,m^{-2}$	18 Jun 2015	12.0a[b]	69.0b	46.9a	34.2a
	10 Sep 2015	5.5a	29.1b	20.7a	13.8a
	06 Jun 2016	10.6a	89.2b	49.7a	50.2a
	07 Sep 2016	6.4a	28.7b	16.8a	18.2a
	Across dates	8.6a	54b	33.5a	29.1a
Standing residue biomass, $g\,m^{-2}$	18 Jun 2015	1.93a	8.69b	5.56a	5.05a
	10 Sep 2015	1.71a	5.71b	4.60a	2.81a
	06 Jun 2016	1.91a	12.69b	7.62a	6.97a
	07 Sep 2016	1.80a	5.15b	3.03a	3.92a
	Across dates	1.84a	8.06b	5.20a	4.69a
Residue stem area index, $m^2\,m^{-2}$	18 Jun 2015	0.002a	0.008b	0.005a	0.005a
	10 Sep 2015	0.001a	0.006b	0.005a	0.003a
	06 Jun 2016	0.002a	0.012b	0.007a	0.007a
	07 Sep 2016	0.002a	0.005b	0.002a	0.005a
	Across dates	0.002a	0.008b	0.005a	0.005a
PM10 fraction (%)	18 Jun 2015	0.81a	1.07a	0.77a	1.11b
	10 Sep 2015	0.97a	1.61a	1.39a	1.19a
	06 Jun 2016	1.25a	0.95b	1.08a	1.13a
	07 Sep 2016	2.23a	1.91a	1.77a	2.37a
	Across dates	1.32a	1.39a	1.25a	1.45a
Residue flat cover, $m^2\,m^{-2}$	18 Jun 2015	0.10a	0.27b	0.19a	0.18a
	10 Sep 2015	0.08a	0.28b	0.16a	0.19a
	06 Jun 2016	0.08a	0.25b	0.16a	0.16a
	07 Sep 2016	0.07a	0.15b	0.11a	0.11a
	Across dates	0.08a	0.24b	0.16a	0.16a
Bulk density, $Mg\,m^{-3}$	18 Jun 2015	1.12a	1.08a	1.11a	1.09a
	10 Sep 2015	1.14a	1.10a	1.13a	1.11a
	06 Jun 2016	1.00a	0.99a	1.01a	0.98a
	07 Sep 2016	1.02a	0.99a	1.02a	0.99a
	Across dates	1.07a	1.04a	1.07a	1.04a
GMD[a], mm	18 Jun 2015	1.04a	0.77a	0.86a	0.95a
	10 Sep 2015	0.39a	0.69a	0.67a	0.41a
	06 Jun 2016	0.72a	1.15b	0.88a	0.99a
	07 Sep 2016	0.29a	0.36a	0.43a	0.22a
	Across dates	0.61a	0.74a	0.71a	0.64a
Penetration resistance (kPa)	18 Jun 2015	2.5a	2.6a	2.7a	2.4a
	10 Sep 2015	4.1a	4.6a	4.6a	4.1a
	06 Jun 2016	11.5a	11.2a	11.2a	11.5a
	07 Sep 2016	13.2a	15.1b	14.2a	14.1a
	Across dates	7.8a	8.4a	8.2a	8.0a
Soil water potential, MPa	18 Jun 2015	−284a	−280a	−278a	−286a
	10 Sep 2015	−142a	−121a	−138a	−125a
	06 Jun 2016	−241a	−242a	−261a	−223a
	07 Sep 2016	−136a	−172a	−165a	−144a
	Across dates	−201a	−204a	−211a	−195a
Allmaras random roughness, mm	18 Jun 2015	12.8a	13.6a	14.2a	12.1a
	10 Sep 2015	12.7a	15.4a	15.2a	13.0a
	06 Jun 2016	12.2a	10.6a	11.8a	10.9a
	07 Sep 2016	9.9a	10.3a	10.5a	9.7a
	Across dates	11.9a	12.5a	12.9a	11.4a
Surface water content, $g\,g^{-1}$	18 Jun 2015	0.010a	0.012b	0.011a	0.011a
	10 Sep 2015	0.017a	0.020a	0.017a	0.020a
	06 Jun 2016	0.011a	0.013a	0.011a	0.014a
	07 Sep 2016	0.018a	0.016a	0.015a	0.019b
	Across dates	0.014 a	0.015a	0.014a	0.016b

[a] GMD is the geometric mean diameter of the aggregate size distribution.
[b] Treatment means followed by same letter indicates no significant difference between disk and undercutter tillage or synthetic and biosolid fertilizer at $P \leq 0.05$.

Table 2
Windblown soil loss as influenced by tillage and fertilizer treatments during 2015 and 2016.

Sample date	Soil loss (g m^{-2})							
	Without abrader				With abrader			
	Tillage		Fertilizer		Tillage		Fertilizer	
	Disk	Under-cutter	Synthetic	Biosolid	Disk	Under-cutter	Synthetic	Biosolid
18 Jun 2015	500a[1]	279b	398a	381a	344a	148b	240a	253a
10 Sep 2015	483a	424a	454a	453a	401a	318a	325a	393a
06 Jun 2016	383a	151b	312a	222a	454a	214b	395a	273b
07 Sep 2016	2539a	1366a	2133a	1771a	2057a	1157a	1739a	1475a
Across dates	976a	555b	824a	707a	814a	459b	675a	598a

[1] Tillage or fertilizer means followed by the same letter on a given date are not significantly different at P ≤ 0.05.

Table 3
Split-block analysis of variance showing P-value for source of variation associated with treatment effects and interaction between treatments. The P-value indicates the level of significance for sources of variation on each of the four dates when wind erosion potential was measured under limited (without abrader) and copious saltation activity (with abrader).

Date	P-value					
	Without abrader			With abrader		
	Tillage	Fertilizer	Interaction	Tillage	Fertilizer	Interaction
18-Jun-15	0.03	0.855	0.213	0.002	0.799	0.515
10-Sep-15	0.534	0.988	0.479	0.083	0.143	0.671
6-Jun-16	0.001	0.111	0.883	0.0001	0.02	0.865
7-Sep-16	0.056	0.509	0.923	0.108	0.61	0.536
Across dates	0.013	0.61	0.777	0.001	0.953	0.638

3.3. Temporal variation in soil loss from tillage and fertilizer treatments

Tillage practices influence wind erosion and dust emissions from WW-SF cropping systems in the PNW. For example, undercutter tillage has been shown to reduce soil loss during high wind events by as much as 65% as compared with disk tillage (Sharratt and Feng, 2009). In addition, fewer tillage operations during summer fallow also result in lower soil loss during high winds (Sharratt et al., 2010). Thus, tillage practices and associated differences in surface characteristics may have influenced soil loss across measurement dates.

Differences in soil loss appeared to be greater among measurement dates than between tillage treatments. For example, averaged across tillage and fertilizer treatments, soil loss was three to nine times greater on 07 Sep 2016 than on the other three measurement dates. In contrast, soil loss averaged across measurement dates was no more than three times greater for disk than undercutter tillage or synthetic versus biosolids fertilizer treatments. The relatively high soil loss on 07 Sep 2016 was surprising considering the occurrence of a 0.2 mm precipitation event the previous day. This precipitation event did not give rise to a soil crust and thus appeared to have little influence on wind erosion. The relatively high soil loss on 07 Sep 2016 was possibly due to an additional rodweeding in 2016 than in 2015 resulting in a seemingly smaller crop residue cover, GMD, and random roughness. In fact, when averaged across tillage or fertilizer treatments, crop residue cover, GMD, and random roughness were at least respectively 33, 40, and 11% smaller on 07 Sep 2016 as compared to the other three measurement dates (Table 1).

We compared soil loss for specific tillage and fertilizer treatment combinations across measurement dates to determine treatment effects associated with trends in soil loss (Fig. 2). The trend for lower soil loss from undercutter versus disk tillage was apparent for both fertilizer treatments in 2015 and 2016 (Fig. 2C, D, G, and H). Averaged across the four measurement dates and fertilizer treatments, soil loss from

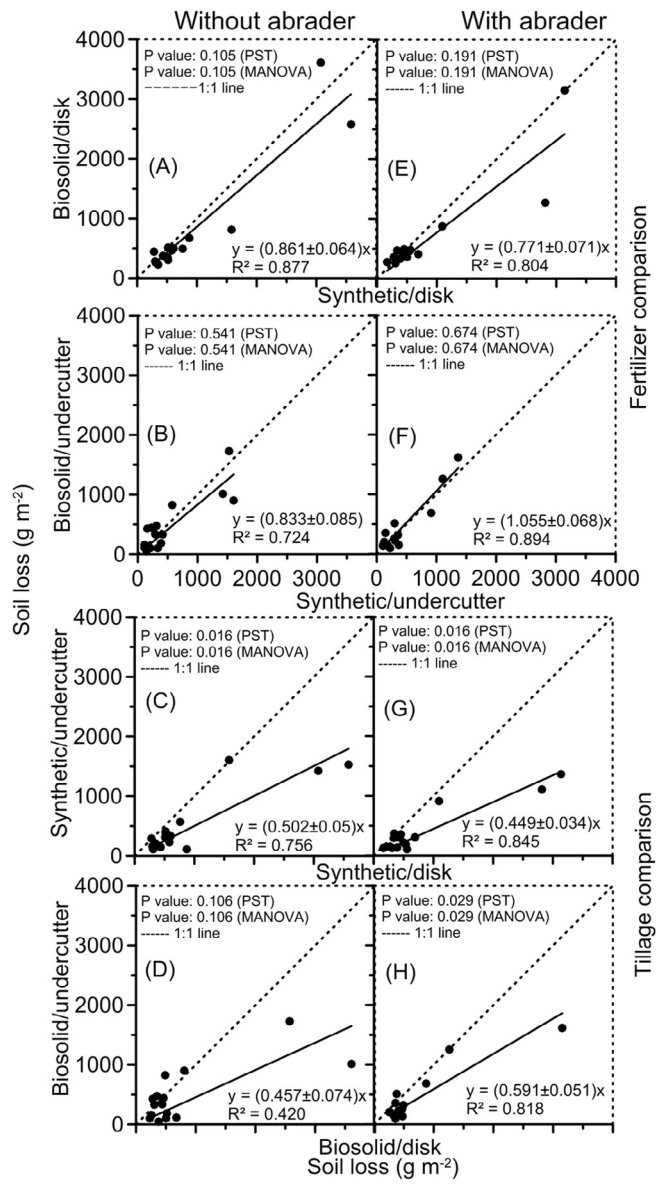

Fig. 2. Comparisons of soil loss among combinations of synthetic and biosolids fertilizer and disk and undercutter tillage treatments across four replications and measurement dates in 2015 and 2016. Soil loss was measured under limited (without abrader) and active saltation (with abrader) conditions using a portable wind tunnel. Multi-factor Analysis of Variance (MANOVA) and paired-samples *t* test (PST) were used to identify significant differences between treatments.

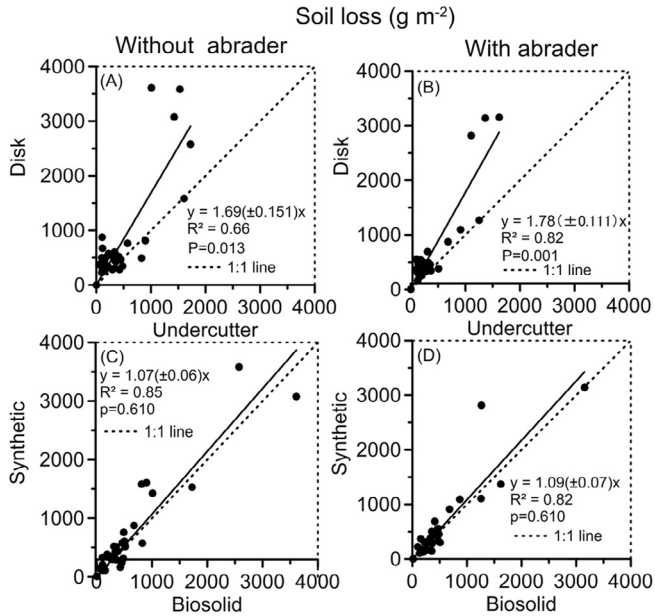

Fig. 3. Comparison of soil loss between synthetic and biosolids fertilizer treatments and disk and undercutter tillage treatments across four replications and measurement dates in 2015 and 2016. Soil loss was measured under limited (without abrader) and active saltation (with abrader) conditions using a portable wind tunnel. A nonparametric test was used to identify significant differences between treatments.

undercutter tillage was about 45% lower than disk tillage (Fig. 3A and B). A nonparametric test also indicated significant differences in soil loss between tillage treatments across dates and fertilizer treatments (Fig. 3A and B). Lower soil loss from undercutter tillage may be due to the higher biomass in undercutter than disk tillage. Indeed, prostrate and standing residue biomass, residue stem area index, and residue flat cover were respectively 528, 338, 300, and 200% higher for undercutter versus disk tillage when averaged across all measurement dates (Table 1). The slope estimates of the relationship between soil loss from disk versus undercutter indicated lower soil loss from undercutter versus disk tillage under more erodible conditions. Thus, treatment effects appear to be dependent on the severity or magnitude of an erosion event. Sharratt et al. (2010) demonstrated the effectiveness of undercutter tillage to reduce wind erosion as compared with traditional tillage in the PNW.

A trend for lower soil loss from biosolids versus synthetic fertilizer was apparent for both tillage treatments in 2015 and 2016 (Fig. 2A, B, and E), except for undercutter tillage under copious saltation activity conditions (Fig. 2F). Soil loss from the biosolids fertilizer treatment was 7 and 8% lower than the synthetic fertilizer treatment under respectively limited and active saltation conditions when averaged across the four measurement dates and tillage treatments (Fig. 3C and D). However, a nonparametric test indicated no significant difference in soil loss between fertilizer treatments across dates and tillage treatments (Fig. 3C and D). Thus, the application of biosolids versus synthetic fertilizer appeared to have little or no impact on wind erosion in this study.

Our observations were taken immediately after the second application of biosolids or four years after the first application of biosolids to the experimental plots. Since soil properties slowly respond to changes in management (Sharratt and Schillinger, 2016; Gollany et al., 2011) even after eight years of diverse crop rotations in this semiarid region (Schillinger et al., 2007), we believe contrasts in wind erosion may emerge with the continued application of fertilizer treatments to the experimental plots.

Our observations suggest that biosolids will have little or no affect on wind erosion. The application of biosolids to semiarid lands has been advocated as a means to recycle or utilize wastewater to sustain human life through food production. The threat for runoff of biosolids from semiarid lands is minimal due to high evaporative demands and limited precipitation in these regions. In addition, the application of biosolids has been promoted for improving soil quality. Thus, the application of biosolids appears to have minimal impact on the environment in semiarid regions.

4. Conclusion

Our observations indicate undercutter tillage significantly reduces wind erosion compared to disk tillage. Furthermore, evidence was found to suggest that wind erosion potential was similar to or lower for biosolids as compared with synthetic fertilizer. The application of biosolids has been promoted for improving soil quality, and while soil carbon content was higher in experimental plots that received biosolids, there is a need to assess the continual impact of biosolids applications on wind erosion to further the sustainability of biosolids applications to agricultural lands.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.aeolia.2018.01.009.

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