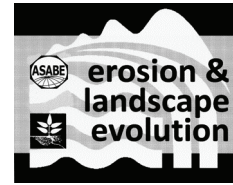# APPLICATION OF THE WIND EROSION PREDICTION SYSTEM IN THE AIRPACT REGIONAL AIR QUALITY MODELING FRAMEWORK

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ABSTRACT. *Wind erosion of soil is a major concern of the agricultural community, as it removes the most fertile part of the soil and thus degrades soil productivity. Furthermore, dust emissions due to wind erosion degrade air quality, reduce visibility, and cause perturbations to regional radiation budgets. PM_{10} emitted from the soil surface can travel hundreds of kilometers downwind before being deposited back to the surface. Thus, it is necessary to address agricultural air pollutant sources within a regional air quality modeling system in order to forecast regional dust storms and to understand the impact of agricultural activities and land-management practices on air quality in a changing climate. The Wind Erosion Prediction System (WEPS) is a new tool in regional air quality modeling for simulating erosion from agricultural fields. WEPS represents a significant improvement, in comparison to existing empirical windblown dust modeling algorithms used for air quality simulations, by using a more process-based modeling approach. This is in contrast with the empirical approaches used in previous models, which could only be used reliably when soil, surface, and ambient conditions are similar to those from which the parameterizations were derived. WEPS was originally intended for soil conservation applications and designed to simulate conditions of a single field over multiple years. In this work, we used the EROSION submodel from WEPS as a PM_{10} emission module for regional modeling by extending it to cover a large region divided into Euclidean grid cells. The new PM_{10} emission module was then employed within a regional weather and chemical transport modeling framework commonly used for comprehensive simulations of a wide range of pollutants to evaluate overall air quality conditions. This framework employs the Weather Research and Forecasting (WRF) weather model along with the Community Multi-scale Air Quality (CMAQ) model to treat ozone, particulate matter, and other air pollutants. To demonstrate the capabilities of the WRF/EROSION/CMAQ dust modeling framework, we present here results from simulations of dust storms that occurred in central and eastern Washington during 4 October 2009 and 26 August 2010. Comparison of model results with observations indicates that the modeling framework performs well in predicting the onset and timing of the dust storms and the spatial extent of their dust plumes. The regional dust modeling framework is able to predict elevated PM_{10} concentrations hundreds of kilometers downwind of erosion source regions associated with the windblown dust, although the magnitude of the PM_{10} concentrations are extremely sensitive to the assumption of surface soil moisture and model wind speeds. Future work will include incorporating the full WEPS model into the regional modeling framework and targeting field measurements to evaluate the modeling framework more extensively.*

Keywords. Air quality, GIS, PM_{10}, Regional modeling, Wind erosion.

W indblown dust is an intermittent but important contributor to detrimental air quality. It can lead to extremely high levels of particulate matter (PM), especially PM_{10} (PM with aerodynamic diameter less than 10 μm), but also $PM_{2.5}$ (PM with aerodynamic diameters less than 2.5 μm) (Sharratt and Lauer, 2006). In the inland Pacific Northwest (PNW), where ~3.3 million ha are used for dryland agriculture, windstorms associated with the passage of intense low-pressure systems periodically cause large dust events that result in significant agricultural soil erosion, produce dangerous driving conditions, and greatly increase atmospheric PM levels (Claiborn et al., 1998; Sundram et al., 2004). Across the U.S. and around the globe, there are many regions where similar conditions exist; examples include Colorado (van Donk and Skidmore, 2003); Mexico

City (Diaz-Nigenda et al., 2010); New South Wales, Australia (Shao et al., 2007); and Alberta, Canada (Coen et al., 2004). Sharratt and Lauer (2006) analyzed ambient PM_{10} data for Kennewick, Washington, and found that the daily PM_{10} concentrations exceeded the U.S. Environmental Protection Agency (EPA) National Ambient Air Quality Standard (NAAQS) (150 µg m^{-3} for 24 h average) 38 times between 1987 and 2005 (16 times between 1999 and 2005). They also reported that 4% to 7% of the PM_{10} measured during these storms was $PM_{2.5}$, suggesting that dust storms can also contribute to exceedances of the 24 h NAAQS for $PM_{2.5}$, which is currently 35 µg m^{-3}. The Columbia Plateau PM_{10} Project (CP^3; http://pnw-winderosion.wsu.edu), initiated in 1993, has investigated the nature of these large dust storms and examined the effects of agricultural practices on windblown dust (Claiborn et al., 1998; Papendick, 2004; Saxton et al., 2000; Chandler et al., 2002, 2004, 2005). These studies have identified conservation practices that have led to reductions in the occurrence of windblown dust in the PNW (Sharratt and Feng, 2009); however, large storm events still occur in the region. For example, a dust storm covering hundreds of square kilometers occurred on 4 October 2009 in the Columbia Plateau region of eastern and central Washington and northern Oregon (fig. 1). Moreover, under future climate change scenarios, temperatures in the region are predicted to increase, and some climate models predict that precipitation will decrease (Salathé et al., 2010). These results suggest that, under the same land-management practices, soil susceptibility to erosion may increase in the future.

An important aspect of understanding the impact of agricultural activities and land-management practices on air quality is the ability to address the effect of dust emissions in the context of other air pollutant sources. This can be done within a regional air quality modeling system that addresses multiple pollutant sources and accounts for meteorological and land-use conditions. Washington State University (WSU) has led an effort to develop, maintain, and
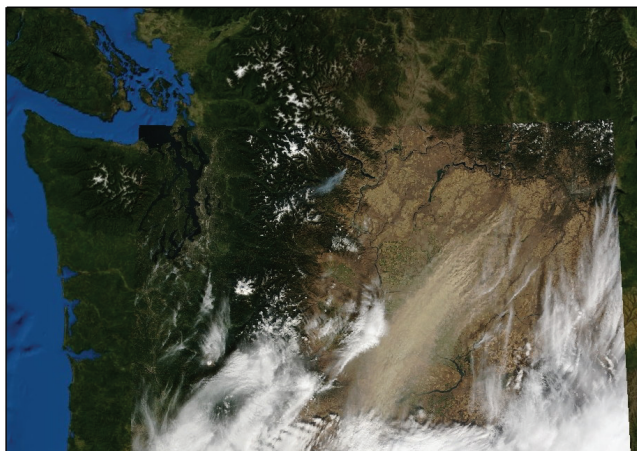

Figure 1. Image captured by the Moderate Resolution Imaging Spectroradiometer (MODIS) on NASA's Terra satellite showing the 4 October 2009 dust event at ~11:00 a.m. PST over the Columbia Plateau region of eastern and central Washington and northern Oregon (http://earthobservatory.nasa.gov/IOTD/view.php?id=40590; NASA image courtesy Jeff Schmaltz, MODIS Rapid Response Team).

improve a comprehensive, automated air quality forecast system for the Pacific Northwest called AIRPACT (Air Indicator Report for Public Awareness and Community Tracking; http://lar.wsu.edu/airpact; Chen et. al., 2008), which is supported by the Northwest International Air Quality and Environmental Science consortium (NW-AIRQUEST; http://lar. wsu.edu/nw-airquest). A variety of industrial, automobile, vegetation, and agriculture sources are incorporated into the AIRPACT framework to address ozone and $PM_{2.5}$, but the air quality impacts of windblown dust on PM_{10} are not currently addressed.

Previous efforts in modeling of windblown dust emissions for the PNW used semi-empirical approaches that did not provide for an objective, systematic treatment of soil physics or land-management practices. In contrast, the Wind Erosion Prediction System (WEPS) treats soil erosion and dust production within a more process-based physical framework. Development of WEPS has been led by the USDA Agricultural Research Service (ARS). It is an upgrade over the earlier empirical Wind Erosion Equation model (WEQ; Woodruff and Siddoway, 1965), which had been the most widely used model for assessing annual windblown soil loss from agricultural fields. WEPS represents a potentially significant improvement in comparison to previous windblown dust modeling algorithms that employed only simple physics and relied heavily on empirical formulations (Gillette, 1988; Gillette and Hanson, 1989; Gillette and Passi, 1988; Nickovic et al., 2001; Shao, 2004; Shaw et al., 2008).

The goal of this study is to incorporate windblown dust within a regional air quality framework that can be utilized for three main purposes: (1) as a warning system for dangerous road conditions, (2) to evaluate the impact of windblown dust on air quality in the context of other pollution sources, and (3) to study the impact of climate change and land-management practices on windblown dust emissions and air quality. To accomplish this, we took advantage of the physical algorithms embedded in WEPS. In this work, WEPS is incorporated into the AIRPACT regional air quality modeling system for the first time. First, we used the EROSION submodel of WEPS as a regional PM_{10} emission module by extending it to cover a large region represented by Euclidean grid cells. With the aid of satellite products to specify field conditions, the new PM_{10} emission module was then employed within AIRPACT, which includes the WRF weather model and the CMAQ chemical transport model that are widely used in the air quality forecasting and research community. For this work, a version of CMAQ was used in which only windblown dust was treated. Once the full WEPS model is incorporated, this framework can be used to study the impact of climate change and land-management practices on windblown dust emissions and air quality in the context of other air pollutant sources. To demonstrate the capabilities of the WRF/EROSION/CMAQ dust modeling system and to determine critical improvements needed for full integration of the WEPS model, we present here results from simulations of dust storms that occurred in central and eastern Washington during 4 October 2009 and 26 August 2010. We also present results from several additional simulations to investigate the sensitivity

of the model results to wind speeds and the spatial scale at which EROSION is run.

DUST EMISSION MODELING AND WEPS

Previous efforts in modeling windblown dust emissions for the Columbia Plateau region of the PNW have been based on a simple dust emission equation (Claiborn et al., 1998) or on the EMIT dust algorithm (Saxton et al., 2000; Sundram et al., 2004). The former of these takes the following form:

$$F = Cu_*^{\alpha}\left(u_* - u_{*_t}\right) \qquad (1)$$

where F is emission flux (g m^{-2} s^{-1}), C is an empirical normalization constant (g m^{-6} s^3 for $\alpha = 3$ and g m^{-5} s^2 for $\alpha = 2$), the exponent α is 3 (e.g., Claiborn et al., 1998) or 2 (e.g., Gillette, 1978), u_* is the friction velocity (m s^{-1}), and u_{*_t} is the threshold friction velocity (m s^{-1}), i.e., the friction velocity above which erosion processes can occur. The friction velocity is defined by equation 2:

$$u = \frac{u_*}{k}\ln\left(\frac{z}{z_0}\right) \qquad (2)$$

where u (m s^{-1}) is the wind speed at height z (m), z_0 (m) is the surface roughness, and $k = 0.4$ is von Karman's constant. The implementation of equation 1 in most earlier studies assumed that u_{*_t} varied with soil class but was static in time, even though in reality u_{*_t} changes with vegetation cover, soil moisture, other surface conditions, and whether or not erosion processes have already been initiated. Most earlier studies also assumed that the parameter C in equation 1 is a constant, even though it changes with surface roughness and vegetation cover. The empirical nature of these earlier approaches did not provide for an objective, systematic treatment of soil physics or land-management practices. In contrast, WEPS treats soil erosion and dust production within a more process-based physical framework. In WEPS, the erosion equation takes the same form as equation 1 with $\alpha = 2$, but WEPS employs explicit time-dependent calculations to determine u_{*_t} and C, accounting for vegetation type, vegetation cover, soil moisture, and soil properties that can be affected by land-management practices (Hagen, 1991). Furthermore, WEPS uses two types of threshold friction velocity. The static threshold friction velocity is the friction velocity above which erosion processes are initiated, while the dynamic threshold friction velocity is the threshold friction velocity above which erosion processes continue. The dynamic threshold friction velocity is less than the static threshold friction velocity because it takes more force to begin moving the soil aggregates than to maintain the movement once it has been initiated. Another improvement over previous models used for the PNW is that WEPS treats direct emissions from both saltation and suspension size aggregates, although the same threshold friction velocity is assumed for both processes. This is especially important in central and eastern Washington, where a large portion of the PM emission can arise from direct suspension (Kjelgaard et al., 2004). A review of various approaches to modeling erosion, including WEPS, can be found in Webb and McGown (2009).

WEPS is a field-based model originally designed to estimate soil erosion and PM$_{10}$ generation for a single field over multiple years. For air quality concerns, windblown dust is a regional problem because PM$_{10}$ emitted from the soil surface can travel hundreds of kilometers downwind before being redeposited, and thus is best addressed within a regional modeling framework. Incorporation of WEPS into a regional air quality framework requires substantial changes to WEPS' programming architecture. To date, this implementation is unique to the state of Washington (Gao et al., 2013). The EROSION submodel of WEPS is the component that simulates the physics of soil erosion and PM$_{10}$ emission; it can be operated as a stand-alone model, typically referred to as SWEEP (Single-event Wind Erosion Evaluation Program) when used with its graphical user interface program. EROSION calculates the threshold friction velocity, accounting for effects of surface soil conditions such as moisture content, roughness, biomass cover, and standing biomass leaf and stem area indices. If the friction velocity exceeds the threshold friction velocity, the EROSION submodel simulates the erosion processes, including saltation, abrasion, suspension, and resulting PM$_{10}$ emissions. While the eventual goal of this work is to utilize the full WEPS model to take advantage of its ability to account for crop growth and land-management practices, discussed by Gao et al. (2013), a concurrent effort is reported here to implement and evaluate only the EROSION submodel in a regional air quality modeling system. This will provide insights into how well WEPS performs on a regional scale in the PNW and thus guide improvements needed for regional air quality applications.

While WEPS is a more process-based model in comparison to previous dust emission algorithms that have been applied to regional air quality modeling, it still contains empirical constants (e.g., the bare soil threshold friction velocity) that have been derived based on soil samples from a finite set of locations. Thus, model evaluation is still an important aspect of our ongoing work. Several studies have applied and evaluated WEPS for various locations in the U.S. and Canada (e.g., Coen et al., 2004; Hagen, 2004), including the Columbia Plateau region that is the focus of this study (Feng and Sharratt, 2007b, 2009). However, most studies have focused on soil loss at the field scale (~10 ha) and near-field (~10 m) impacts. Feng and Sharratt (2007a) used WEPS to predict annual soil loss and PM$_{10}$ emissions for Adams County, Washington, but the air quality impacts of PM$_{10}$ downwind of the source region were not investigated. As figure 1 indicates, soil erosion in the Columbia Plateau region can adversely affect air quality more than 100 km downwind. The only study that has applied the EROSION submodel in a regional air quality modeling framework is that of Diaz-Nigenda et al. (2010), who applied it to Mexico City.

METHODS

The modeling framework used in this study is very similar to the AIRPACT-3 air quality modeling system for the PNW. Figure 2 shows a schematic of how the EROSION submodel is incorporated into this air quality modeling framework, which uses the WRF (Weather Research and Forecasting; http://wrf-model.org/index.php; Skamarock et al., 2008) meteorology model and the CMAQ (Community Multiscale Air Quality; www.cmascenter.org; Byun and Schere, 2006) chemical transport model. The AIRPACT-3 simulation domain encompasses Washington, Oregon, Idaho, and bordering areas (domain 1 in fig. 3). In forecast mode, the simulation domain is divided into 95 × 95 Euclidean cells of 12 km × 12 km horizontal dimension using the Lambert conformal conic projection. For simplicity, we will refer to this domain as a "12 km grid" and the individual 12 km × 12 km cells as "12 km cells"; grids and cells of other sizes are denoted similarly. Vertically, there are 21 layers, extending to the 100 mb pressure level, with the bottom layer comprising the first ~35 m above the surface. Each day, AIRPACT-3 provides an automated 64 h air quality forecast beginning at 08:00 GMT (midnight PST) for the next day. The system is initiated daily at midnight local time and is able to complete the entire simulation and post-processing to make graphical output available on a public web page by ~4:00 a.m. local time (http://lar.wsu.edu/airpact). Although AIRPACT-3 is automated for forecast mode, it is straightforward to perform retrospective analyses for historical cases, as described in this article. For this study, the analysis focuses on the Columbia Plateau region, which consists of central and eastern Washington and northern Oregon. In addition to AIRPACT-3, the WRF-

CMAQ modeling framework is widely used to study the impact of various pollution sources on regional air quality (e.g., Carlton et al., 2010) and to study the impact of climate change in air quality (e.g., Avise et al., 2012); thus, progress in demonstrating the value of WEPS within this context could have wider applicability.

WINDBLOWN DUST PM$_{10}$ EMISSION MODELING: THE EROSION SUBMODEL

WEPS was originally designed to simulate the temporal variability of field conditions and the soil loss/deposition and PM$_{10}$ emissions within a single field over time. In this work, rather than simulating a single field, we run EROSION on multiple cells independently. We also modified the EROSION submodel to use hourly wind directions from the WRF model instead of daily wind directions. Eulerian regional air quality models typically subdivide the simulation domain into a three-dimensional array of Euclidean cells (volumes). The horizontal cell dimensions typically range between 4 km × 4 km and 36 km × 36 km, depending on the size of the simulation domain and computational limitations (time and memory), although sometimes 1 km × 1 km cells are used for urban areas. In this study, we use three nested simulation domains with WRF and CMAQ, using 12 km, 4 km, and 1 km cells (fig. 3). The larger 12 km cell spacing allows for simulations of a larger domain, which is necessary when windblown dust can affect air quality more than 100 km downwind from the source. At 1 km cell spacing, the spatial scale is closer to the original design of WEPS, and the assumptions of uniform soil, surface, and ambient conditions within each cell are less problematic. However, using 1 km cells also requires the simulation domain to be smaller to
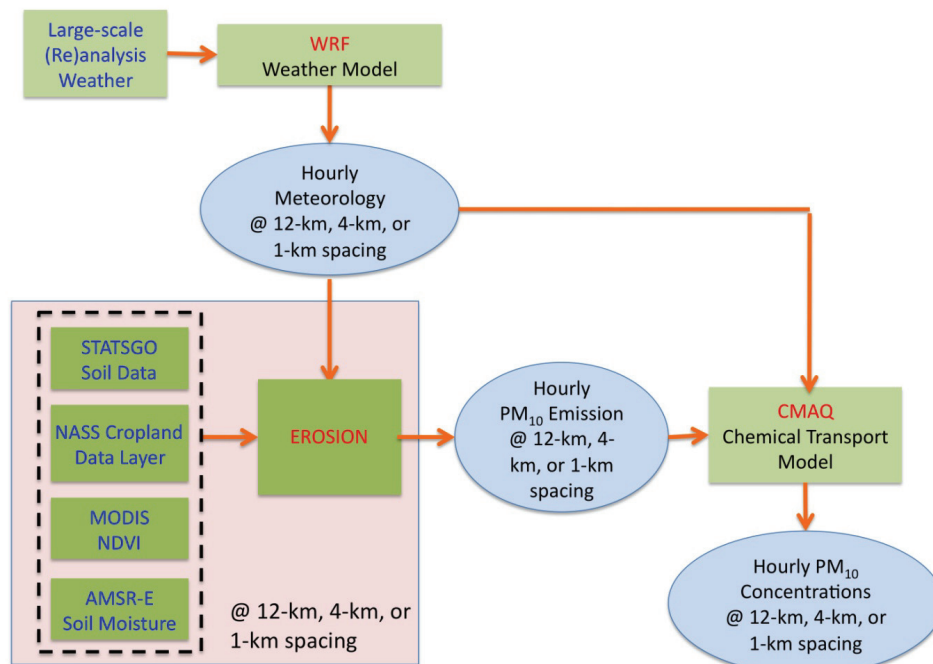
Figure 2. Flow diagram for incorporation of the WEPS EROSION submodel into a WRF-CMAQ air quality modeling framework: WRF = Weather Research and Forecasting, CMAQ = Community Multiscale Air Quality, STATSGO = State Soil Geographic, NASS = National Agricultural Statistics Service, MODIS = Moderate Resolution Imaging Spectroradiometer, NDVI = normalized difference vegetation index, and AMSR-E = Advanced Microwave Scanning Radiometer for EOS (Earth Observing System).

Figure 3. WRF-EROSION-CMAQ simulation domains and cropland fraction derived from the 2010 Cropland Data Layer for the U.S. (www.nass.usda.gov/research/Cropland/SARS1a.htm) and the 2000 G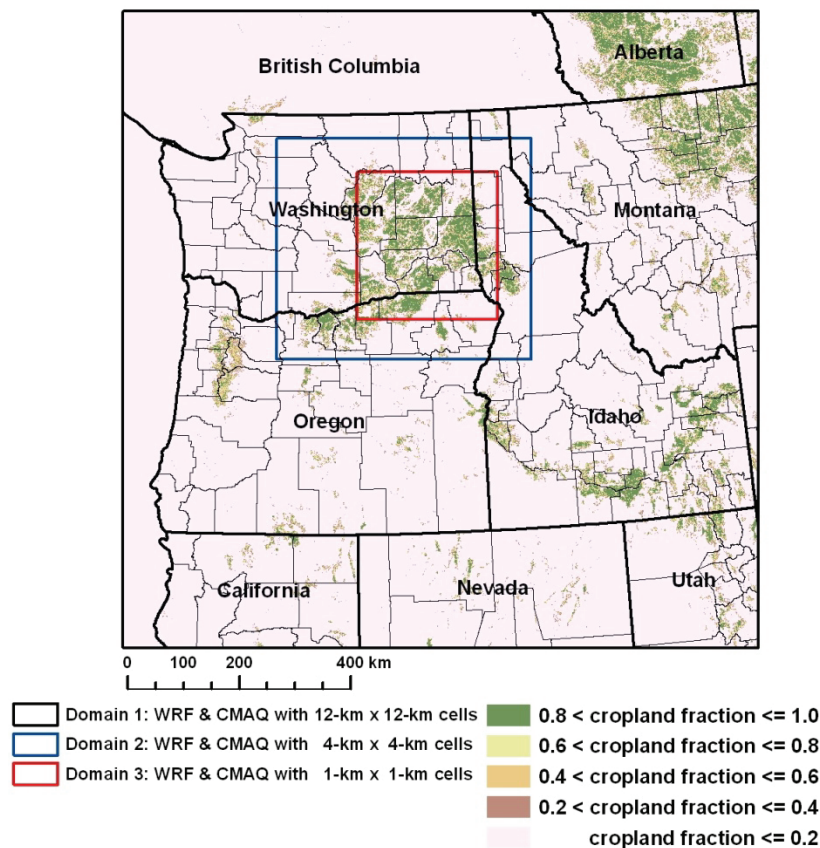eoBase land cover dataset for Canada (www.geobase.ca/geobase/en/data/landcover/index.html). Cropland fraction is defined as the percentage of the area in each 1 km × 1 km cell that is cropland.

reduce computational cost, which is especially critical for forecasting applications for which simulations must finish before the events happen. To illustrate the tradeoffs between grid setup and computational expense, a 36 h WRF simulation for domain 1 using 12 km cells requires ~1 h of computing time using eight 2.3 GHz central processing units on our cluster; however, simulating the same domain using 1 km cells would require more than 100 h. For the same grid setup, similar computational time is required to run CMAQ with only one non-reactive PM_{10} category.

In this study, we evaluate the modeling system using various combinations of 12 km, 4 km, and 1 km cell spacings for EROSION and WRF/CMAQ to determine a suitable combination. For each WRF/CMAQ cell, we run EROSION either on the same cells or on smaller cells created by further subdividing the WRF/CMAQ cells into smaller cells (i.e., a 12 km cell might be subdivided into nine 4 km cells or 144 one-kilometer cells, and a 4 km cell might be subdivided into sixteen 1 km cells). When the WRF/CMAQ 12 km or 4 km cells were subdivided into 1 km cells for running EROSION, 12 km, 4 km, and 1 km wind fields from WRF were uniformly applied to the 1 km EROSION cells; the impact of using different WRF cell sizes on 1 km EROSION runs is discussed in the Results and Discussion section. Within each EROSION cell, we run EROSION up to eight times, depending on the number of soil map units found within that cell as determined by the STATSGO soil data (see below). For each soil map unit

within an EROSION cell, we run EROSION assuming a field the size of the cell with uniform soil properties and uniform field and meteorological conditions. The total PM_{10} emission rate in each EROSION cell is then calculated as the mean of the area-weighted emission rates for each soil map unit. For input into the CMAQ model, we summed the total PM_{10} emissions from all EROSION cells contained within each CMAQ cell. This is repeated for the entire modeling domain for each time step for each simulation.

Input Data for the EROSION Submodel

The four principal data sets characterizing the surface that are required to run the regional-scale EROSION submodel are land use, soil properties, soil moisture, and crop cover data. These data describe the conditions of the surface that are necessary for EROSION to calculate the threshold friction velocity above which erosion occurs. The soil moisture and crop cover data, in particular, are needed because we are not using the other components of the WEPS model. We make use of satellite products to estimate these required input parameters for the EROSION submodel. All the satellite products discussed in the following paragraphs are mapped using Geophysical Information System (GIS) analysis onto the 1 km, 4 km, and 12 km cells of our AIRPACT-3 domain and then used by EROSION at these respective cell spacings. Table 1 summarizes the values and references for the input parameters used for the EROSION simulations.

Table 1. Input parameters for the EROSION simulations.

Parameter	Values or Source
Field dimension (m)	EROSION grid cell size
Biomass height (m)	0 for MODIS NDVI <1.5
Stem area index ($m^2 m^{-2}$)	0 for MODIS NDVI <1.5
Leaf area index ($m^2 m^{-2}$)	0 for MODIS NDVI <1.5
Biomass flat cover ($m^2 m^{-2}$)	0 for MODIS NDVI <1.5
Bulk density ($Mg m^{-3}$)	Aggregated values from STATSGO (Feng et al. 2009)
Sand content ($kg kg^{-1}$)	Aggregated values from STATSGO (Feng et al. 2009)
Very fine sand ($kg kg^{-1}$)	Aggregated values from STATSGO (Feng et al. (2009)
Silt content ($kg kg^{-1}$)	Aggregated values from STATSGO (Feng et al. 2009)
Clay content ($kg kg^{-1}$)	Aggregated values from STATSGO (Feng et al. 2009)
Rock volume fraction ($m^3 m^{-3}$)	0
Aggregate density ($Mg m^{-3}$)	Aggregated values from STATSGO (Feng et al. 2009)
Aggregate stability ($\ln[J kg^{-1}]$)	Aggregated values from STATSGO (Feng et al. 2009)
Aggregate geometric diameter (mm)	Aggregated values from STATSGO (Feng et al. 2009)
Minimum aggregate size (mm)	0.01
Maximum aggregate size (mm)	Aggregated values from STATSGO (Feng et al. 2009)
Aggregate geometric standard deviation ($mm mm^{-1}$)	Aggregated values from STATSGO (Feng et al. 2009)
Fraction of soil surface crusted ($m^2 m^{-2}$)	0
Soil crust thickness (mm)	0
Fraction of crusted surface covered by loose material ($m^2 m^{-2}$)	0
Mass of loose material on crusted surface ($kg m^{-2}$)	0
Soil crust density ($Mg m^{-3}$)	1.7
Soil crust stability ($\ln[J kg^{-1}]$)	1.35
Random roughness (mm)	10
Soil wilting point water content ($kg kg^{-1}$)	Aggregated values from STATSGO (Feng et al. 2009)
Surface water content ($kg kg^{-1}$)	See text
Ridge height (mm)	N/A
Ridge space (mm)	N/A
Ridge orientation (°)	N/A
Wind speed ($m s^{-1}$)	WRF model
Wind direction (°)	WRF model

We run the EROSION submodel only for cells that contain cropland. The USDA-NASS Cropland Data Layer (CDL) (www.nass.usda.gov/research/Cropland/SARS1a.htm) for 2010 is used to identify the percentage area of cropland within each cell (whether 1 km, 4 km, or 12 km). The CDL is a raster, geo-referenced, land-cover dataset with a ground resolution of 30 m for 2010. The data are generated using multiple satellite images, including the Landsat 5 TM sensor, Landsat 7 ETM+ sensor, and the Indian Remote Sensing RESOURCESAT-1 (IRS-P6) Advanced Wide Field Sensor (AWiFS) during each respective growing season. To improve the classification, The CDL also uses ancillary inputs from the USGS National Elevation Dataset (NED), the USGS National Land Cover Database 2001 (NLCD 2001), and the MODIS 250 m 16-day normalized difference vegetation index (NDVI) composites. The CDL provides information on 64 different crop types for Washington. Since we are using the CDL only to identify the percentage area in each grid cell that is cropland and not to identify specific crop types, we can apply the 2010 CDL data to 2009 as well. In contrast, the year-specific

crop type information is used when incorporating the full WEPS model into the regional modeling framework (Gao et al., 2013).

For soil property inputs to EROSION, we use the multi-scale database of Feng et al. (2009). Feng et al. (2009) spatially mapped existing soil properties from 19,681 soil map units in the USDA-NRCS State Soil Geographic (STATSGO) database onto the AIRPACT-3 domain and then estimated soil properties based on quantitative relationships among existing soil properties. Each map unit in the aggregated database is defined by ten soil layers, with each layer characterized by 31 soil physical, chemical, and hydraulic properties derived from aggregating properties of individual soil components. Feng et al. (2009) had mapped the data to 1 km and 12 km cells in the AIRPACT-3 domain; in this study, we additionally mapped the data to 4 km cells in order to evaluate the sensitivity of model results to the spatial scale at which EROSION is run. Each cell can contain multiple soil map units. In our simulations, we run EROSION for each map unit, up to eight map units per cell, by first assuming the entire cell is covered by that map unit and then weighting the resulting PM_{10} emissions by the fractional area of the map unit in the cell.

The amount of vegetative and residue material above the soil surface is an important factor in determining the susceptibility to wind erosion. In EROSION, the cover above the soil surface is characterized by variables such as the growing crop leaf area index and the residue leaf area index. In field-scale simulations, these parameters are estimated by running the CROP submodel in WEPS to simulate crop growth and the MANAGEMENT submodel to simulate tillage, planting, and harvesting. Since the full WEPS model has not yet been adapted to regional-scale simulation, in this work we instead make use of the MODIS 1 km 16-day NDVI (products MOD13A2 and MYD14A2 from the Terra and Aqua satellites, respectively) to identify cells that are susceptible to wind erosion. An NDVI value of less than 0.1 typically indicates bare soil (http://earthobservatory.nasa.gov/Features/MeasuringVegetation/). Since MODIS NDVI is based on area-weighted means and we are using the data for grid cells ranging in scales from 1 km to 12 km, we took the conservative approach and treated any cell with cropland area greater than 0 (as determined from 2010 CDL) and MODIS NDVI less than 0.15 as if it were bare soil. The MODIS NDVI data are reported every eight days. In this study, we used the data from the most recent report prior to a storm event, i.e., 30 September 2009 data for the 4 October 2009 event and 21 August 2010 data for the 26 August 2010 event.

Surface soil moisture is another important parameter in determining the susceptibility of the soil to wind erosion. In the full WEPS model, the HYDROLOGY submodel calculates soil moisture. In this work, we used two alternative approaches to specify the surface soil moisture: (1) assuming a fixed surface soil moisture value, or (2) using the daily Level-3 soil moisture product from the Advanced Microwave Scanning Radiometer for EOS (AMSR-E). For the first option, we ran several simulations at different surface soil moisture levels for the 26 August 2010 event to document the sensitivity of modeling results to this parameter.

While the AMSR-E is useful in providing the general trend in surface soil moisture, it has weaknesses for our application. First, the AMSR-E daily Level-3 product is provided on a 25 km grid, much coarser than our grid. Second, the soil moisture in this satellite product refers to the top ~1 cm of the soil (see http://nsidc.org/data/docs/daac/ae_land3_l3_soil_moisture.gd.html), whereas the most critical parameter for erosion is soil moisture in the top few millimeters of the soil. Despite the limitation of the AMSR-E soil moisture product, it is the only dataset of large spatial coverage for our study domain and time periods.

In addition to the parameters discussed above, there are several other input parameters needed by the EROSION submodel (table 1). Several simulations were performed while varying these parameters over the range provided by Feng and Sharratt (2005), which were based on field measurements. The simulation results are relatively insensitive to these parameters and thus are not shown. This is consistent with the results of Feng and Sharratt (2005), who used a combined method of Latin hypercube sampling (LHS) and one-factor-at-a-time (OAT) parameter examination to evaluate the sensitivity of parameters in EROSION in simulating total soil loss, creep/saltation, suspension, and PM_{10} emission. Table 1 summarizes the values or sources for all inputs into the EROSION submodel used in this study.

Static Bare-Soil Threshold Friction Velocity of the EROSION Submodel

The 26 August 2010 dust event resulted in elevated ambient PM_{10} concentrations across the Columbia Plateau region (see fig. 10). However, our original simulation predicted almost no erosion in the region during this period, even when assuming bare soil and extremely low surface soil water content (0.01 g g^{-1}), and even when the modeled wind speeds were artificially increased by a factor of 1.5. Further investigation revealed that the minimum bare-surface static threshold friction (u_{*tb}) in WEPS is ~0.5 m s^{-1} for the surface roughness values of 8 to 12 mm reported by Feng and Sharratt (2009) and size distribution of the aggregates based on the STATSGO database. However, the measured value for this parameter in the Columbia Plateau region can be 0.3 m s^{-1} or lower (Feng and Sharratt, 2007b; Sharratt and Vaddella, 2012). For this reason, we modified the WEPS code to set the bare-surface static threshold friction velocity to 0.3 m s^{-1} for all results presented in this article.

METEOROLOGY MODELING: THE WEATHER FORECASTING AND RESEARCH (WRF) MODEL

The meteorological data used in WEPS/EROSION include hourly wind information and daily weather data. The weather data include attributes of daily precipitation, maximum and minimum temperature, solar radiation, and dew point temperature (Hagen, 1991). The hourly wind data include wind speed and daily direction. In the original field-scale version of WEPS, the daily weather data are typically generated with the CLIGEN generator via simulation using monthly statistical station parameters derived from measured meteorological data. The hourly wind speeds and daily wind directions are usually generated with the WINDGEN generator, again using the historical statistical meteorological variables (van Donk et al., 2005), although historical or other weather and wind data can also be used to drive WEPS. The spatial distribution of CLIGEN and WINDGEN stations is relatively sparse in comparison to the 12 km or 4 km grid cells typically used for regional air quality simulations. In addition to the spatial sparseness of CLIGEN and WINDGEN stations, these data are statistical records, so the generated values have no relationship with any given day's actual weather and thus cannot be used to predict future storm events on a daily basis.

To use WEPS/EROSION within a regional air quality model, we use the mesoscale WRF model to generate meteorological fields necessary to drive the EROSION submodel as well as the CMAQ chemical transport model, instead of using CLIGEN and WINDGEN. We also modified the EROSION submodel to use hourly instead of daily wind directions. WRF is an operational weather forecasting model that is designed to be computationally efficient and flexible for scientific research. At the University of Washington, Mass et al. (2003) employed WRF to provide numerical weather forecasts on a daily basis for the PNW (www.atmos.washington.edu/mm5rt) and to provide meteorological results required by our automated AIRPACT-3 system for air quality forecasting. For AIRPACT-3's daily forecasting operations, the WRF simulations use National Centers for Environmental Prediction (NCEP) Global Forecasting System (GFS) large-scale meteorological fields on a 1° × 1° (~100 km × 100 km) spacing (www.nco.ncep.noaa.gov/pmb/products/gfs) for initial and boundary conditions. As discussed previously, the modeling framework can also be applied for more detailed historical case studies. For the retrospective simulations performed for this study, WRF simulations use the NCEP North America Regional Reanalysis (NARR; www.emc.ncep.noaa.gov/ mmb/rreanl) meteorological fields at ~32 km × 32 km spacing for initial and boundary conditions.

As will be shown in the Results and Discussion section, the WRF model underpredicts the peak wind speeds during the windstorms. Therefore, we have included EROSION simulations for domain 3 using 1 km and 4 km gridded WRF wind speeds that have been modified based on observed wind speeds. For each simulation cell that has one or more observational sites located within the cell, the hourly modeled wind speeds are replaced with the hourly observed values. For each cell that does not have an observational site within it but has one within 10 km of the cell center, the hourly modeled wind speeds are adjusted by the hourly ratio of the observed wind speed at the closest observational site to the modeled wind speed at the grid cell containing that site. For grid cells that do not have an observational site within 10 km of the cell center, the modeled wind speeds are left unchanged.

CHEMISTRY AND TRANSPORT: THE CMAQ CHEMICAL TRANSPORT MODEL

CMAQ is the Community Multi-Scale Air Quality model developed by the U.S. EPA as a comprehensive chemical

transport model with explicit treatment of photochemical gas and aerosol chemistry, transport, and wet and dry deposition, including aqueous-phase cloud processes (Byun and Schere, 2006). CMAQ has been widely used by research and regulatory communities to study contributions of various pollution sources to regional air quality and their impact on public health (e.g., Fann et al., 2012; Dennis et al., 2008), including the impact of climate change on those effects (e.g., Jackson et al., 2010; Tagaris et al., 2009; Chen et al., 2008). For this work, CMAQ has been modified to include windblown PM_{10} particles as an inert tracer, with PM_{10} emission rates from the EROSION submodel used as input to CMAQ. The dry deposition velocity of these PM_{10} particles is assumed to be 0.1 cm s^{-1} (Sundram et al., 2004). This work focuses only on windblown dust; thus, the photochemical processing of gas-phase pollutants and other PM species are not simulated. Eventually, we plan to include windblown dust within our normal full photochemical modeling simulations and forecasts. Since airborne dust provides surfaces for heterogeneous chemical reactions and undergoes coagulation with other particles, its inclusion in the model is expected to have secondary air quality impacts with other pollutants in the model.

Figure 4. Locations of the wind and PM_{10} observation sites used in this study.

RESULTS AND DISCUSSION

In this study, we focus on two recent dust storms, which occurred on 26 August 2010 and 4 October 2009. The 26 August 2010 event was a typical dust storm for the region in that it was associated with westerly and southwesterly winds. This event was particularly useful for detailed analysis because the storm caused elevated PM_{10} concentrations that were recorded at six locations in the Columbia Plateau region. The 4 October 2009 event was so strong that the dust plume was detected by the MODIS sensors on NASA's Aqua and Terra (fig. 1) satellites. This event was rather unusual for the region in that it was associated with northeasterly winds. Figure 4 shows the locations where model results are compared against observations.

THE 26 AUGUST 2010 EVENT

Wind Speed and Direction

A critical factor in being able to predict wind erosion accurately is to have a reliable forecast of wind speeds. In the new regional modeling framework, EROSION calculates the friction velocity using WRF-modeled wind speeds. Erosion only occurs if the friction velocity exceeds a threshold value, with the PM_{10} emission rate being proportional to the cube of the exceedance. Figure 5 shows a comparison of modeled and observed hourly wind speeds at six sites within or near the cells from which the WRF-EROSION model predicted erosion during the 26 August 2010 event. In theory, WRF should perform better when smaller cells are used to better represent the heterogeneity in surface roughness and terrain height. For all cell sizes applied in this study, WRF was able to predict the timing of peak wind speed in the late afternoon at Lind, Ritzville, Eby, and Pendleton, indicating that the model correctly predicted the onset of the windstorm; however, the modeled peak wind speeds
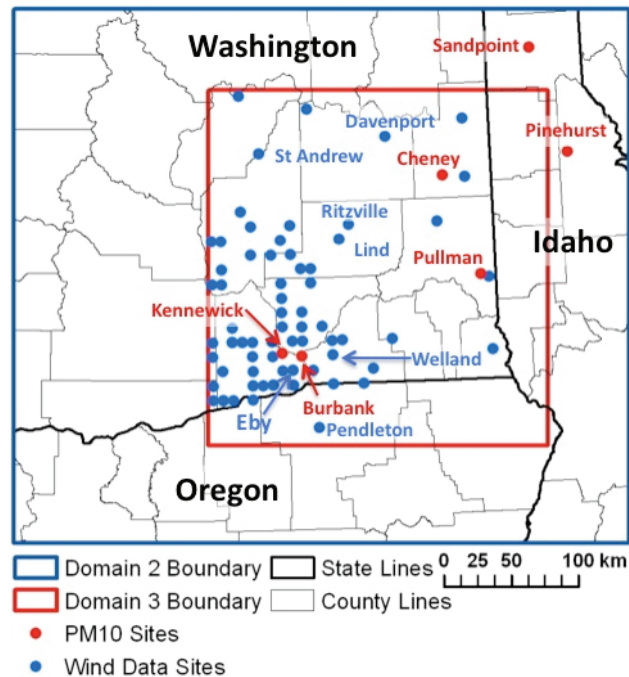
were only ~60% of the observed values, regardless of the cell size used. At Welland and St. Andrews, the model did not predict the afternoon peak in wind speed that was observed. With the exception of the Pendleton location, the choice of model cell size had only minimal impact on the predicted wind speeds. Since the model uniformly underestimated maximum wind speeds throughout the region and for cell sizes ranging from 12 km down to 1 km, it appears that there is a fundamental and systematic bias in the WRF model. It is possible that the model underestimated the regional pressure gradients, or that the boundary layer scheme employs a surface roughness that is too high and produces too much surface drag. However, while the WRF model underpredicted peak wind speeds, there were also instances outside the peak period when the model overpredicted the observed wind speeds, especially when 12 km cells were used. This could be related to timing errors in simulation of the passage of frontal systems.

Figure 6 shows wind rose plots for the aggregate modeled and observed winds for 74 AgWeatherNet stations in domain 3 (fig. 4). Both the observations and model results indicate that winds were predominantly westerly and southwesterly. On average, 12 km WRF predicted higher wind speeds than 4 km WRF, which predicted slightly higher wind speeds than 1 km WRF. The 12 km WRF predicted more westerly than southwesterly wind, whereas the observations and finer-scale WRF simulations all indicated the winds were distributed westerly and southwesterly 30% of the time each. Overall, results for the WRF model with 4 km or 1 km cells showed better correlation with observations for the 74 AgWeatherNet sites within domain 3 for the 26 August 2010 event, both in terms of wind speed and direction (result not shown).
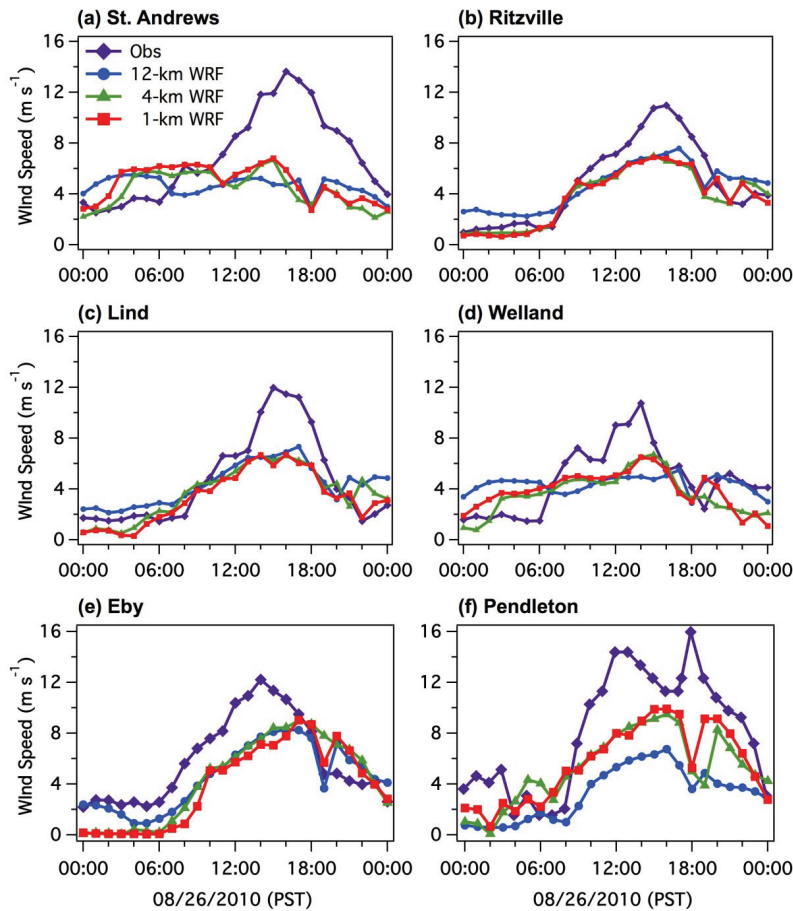
Figure 5. Observed and modeled hourly wind speeds at six sites; locations of the sites are shown in figure 4. The observed wind speeds are reported at 10 m height for Pendleton and at 1.5 to 1.8 m height for the other sites. The modeled wind speeds are reported at 10 m for Pendleton and at 1.8 m for the other sites. The data for Pendleton are from the National Weather Service and obtained via MesoWest (http://mesowest.utah.edu); wind data for the other sites are from WSU's AgWeatherNet (http://weather.wsu.edu).

PM_{10} Emissions

Figures 7 and 8 show the predicted PM_{10} emission rates using various cell sizes for the WRF model and EROSION submodel and surface soil moisture assumptions. Since the model underpredicted peak wind speeds, we also included EROSION simulations based on 1 km and 4 km WRF wind speeds that were adjusted to the observed values for cells within 10 km of an observation station. Each cell used in regional air quality modeling can include considerable heterogeneity in surface soil properties and land use. There were a number of locations in which EROSION was applied using the 12 km cell size and did not predict any erosion, but these same locations showed erosion for simulations using smaller cells. This occurred because we considered each cell to be erodible only if the cell area-mean NDVI was greater than 0.15. If any area of a 12 km cell had NDVI values substantially greater than 0.15, the area-mean NDVI would be greater than 0.15 and the whole cell was considered non-erodible even if part of the cell did in fact have NDVI less than 0.1 and thus should be considered susceptible to erosion. For this same reason, when the EROSION cell size decreased from 4 km to 1 km, predicted PM_{10} emission rates increased by an order of magnitude. Thus, even without consideration of other surface soil conditions, running the EROSION model at finer spatial scales

is important to account for the heterogeneity of biomass cover on the land surface.

In addition to the PM_{10} results being sensitive to the cell size at which WRF and EROSION were run, we found that the predicted PM_{10} emission rates were also very sensitive to assumptions related to surface soil moisture. We ran EROSION with surface soil moistures between 1% and 6% by weight, the range reported for the region from surface measurements during this time of the year (Feng and Sharratt, 2005). Other measurements show that soil water content at 50 mm depth varies by ~1% diurnally during periods with no rain in the region (B. Sharratt, personal communication). Figure 7 indicates that for surface moisture content over the range of 1% to 6%, the predicted total PM_{10} emission rates vary by an order of magnitude, although the range is smaller at higher wind speeds. Figures 8a and 8b show the spatial distribution of the predicted PM_{10} emission rates for surface soil moisture set to 0.01 and 0.06 g g^{-1}, respectively. The reduction in total PM_{10} emissions rates is not only due to reduced rates at the same locations, but also because the spatial extent of where erosion occurs is reduced when surface soil moisture is uniformly increased across the whole simulation domain. For this period, the AMSR-E reported that soil moisture levels were ~0.1 g g^{-1}, which was higher than the range suggested by Feng and
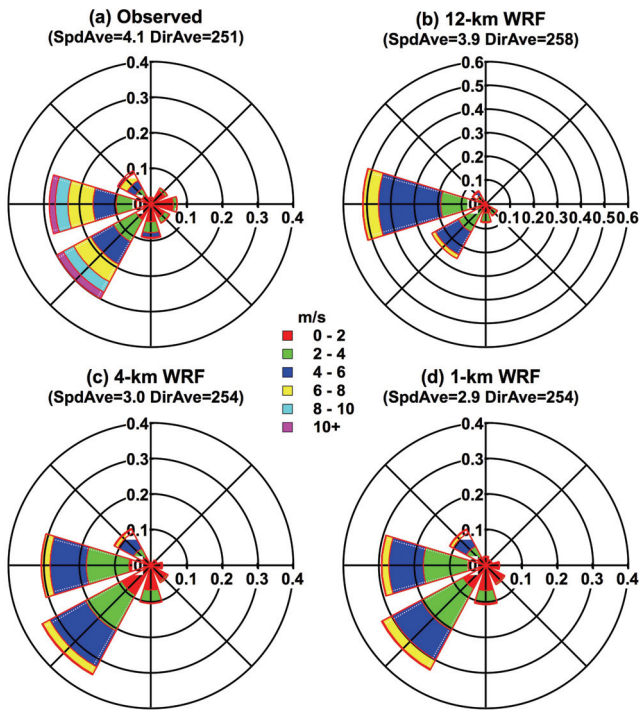
Figure 6. Wind rose plots based on hourly wind fields for 74 Ag-WeatherNet sites (http://weather.wsu.edu) and corresponding WRF cells in domain 3 for 26 August 2010: (a) observation, (b) 12 km WRF, (c) 4 km WRF, and (d) 1 km WRF.

Sharratt (2005) and resulted in reduced PM_{10} emission predictions (fig. 7). As indicated earlier in the Methods section, the AMSR-E soil moisture product has a very coarse spatial scale of 25 km and corresponds to the top ~1 cm of the soil, which is on average more moist than the top 1 mm of the soil that is most critical for erosion. Thus, we conclude that using AMSR-E soil moisture directly leads to

underestimation of PM_{10} emissions. This means that the AMSR-E soil moisture product should not be used directly as input for the EROSION model; however, the temporal and spatial trend in soil moisture provided by the AMSR-E may still be useful as a proxy for top of the surface soil moisture. This is potentially useful given the lack of top of the surface soil moisture data with large spatial coverage at high spatial resolution.

Figure 7 shows that the model results did not change much when the cell size of WRF was changed from 1 km to 12 km while the EROSION simulations used 1 km cells; this was in contrast to the cases in which large differences were found when either the surface soil moisture or the EROSION model cell size was modified. Because the predicted wind speed increased slightly from 1 km and 4 km WRF to 12 km WRF (fig. 6), the predicted PM_{10} emissions also increased. With EROSION running with 1 km cells, PM_{10} emissions increased by 30% to 200% when the WRF cells were increased from 1 km to 4 km and then by another 23% to 43% when the WRF cells were increased to 12 km. These results suggest that using relatively large grid cells for meteorological modeling (while still using smaller grid cells for running EROSION) introduces an uncertainty factor of ~2 in the modeled PM_{10} emission rates for our case study.

As indicated in the previous section, the WRF model underpredicted peak wind speed by up to ~60% regardless of whether 1 km, 4 km, or 12 km cell sizes were used (fig. 5). Thus, we also performed EROSION simulations on domains 2 and 3 for which the 4 km and 1 km WRF wind speeds, respectively, were adjusted using observed wind speeds at the wind data sites shown in figure 4. A summary of the predicted domain-total PM_{10} emission rates is shown in figure 7; and figures 8c and 8d show the spatial distribu-
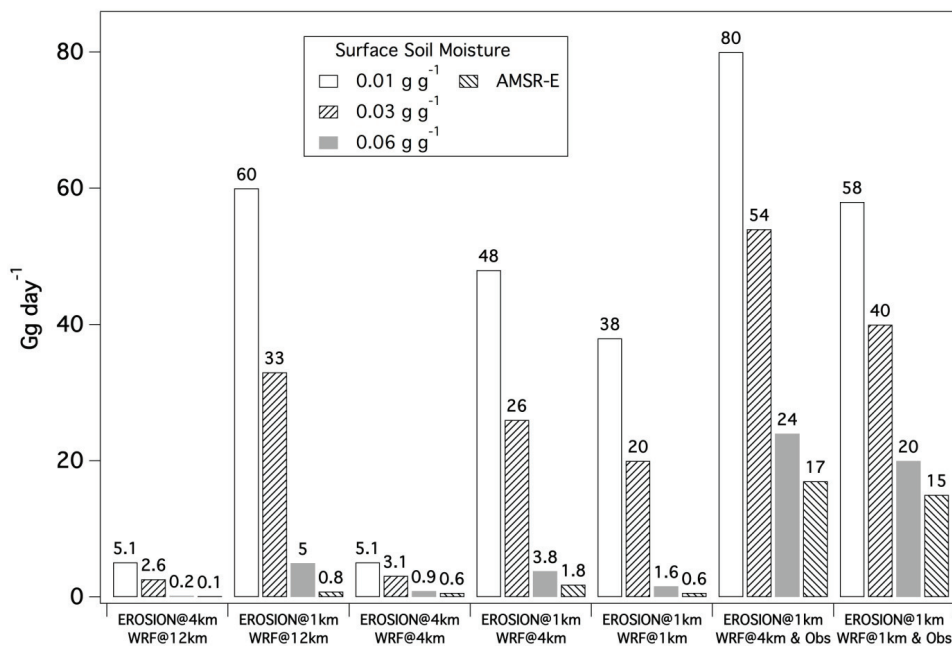

Figure 7. Modeled PM_{10} emission rates for the 26 August 2010 event summed over domain 3. The *x*-axis labels indicate the cell sizes used in the WRF and EROSION simulations for each set of four simulations; "Obs" indicates that the WRF wind speeds were adjusted according to observed wind speeds (see text).
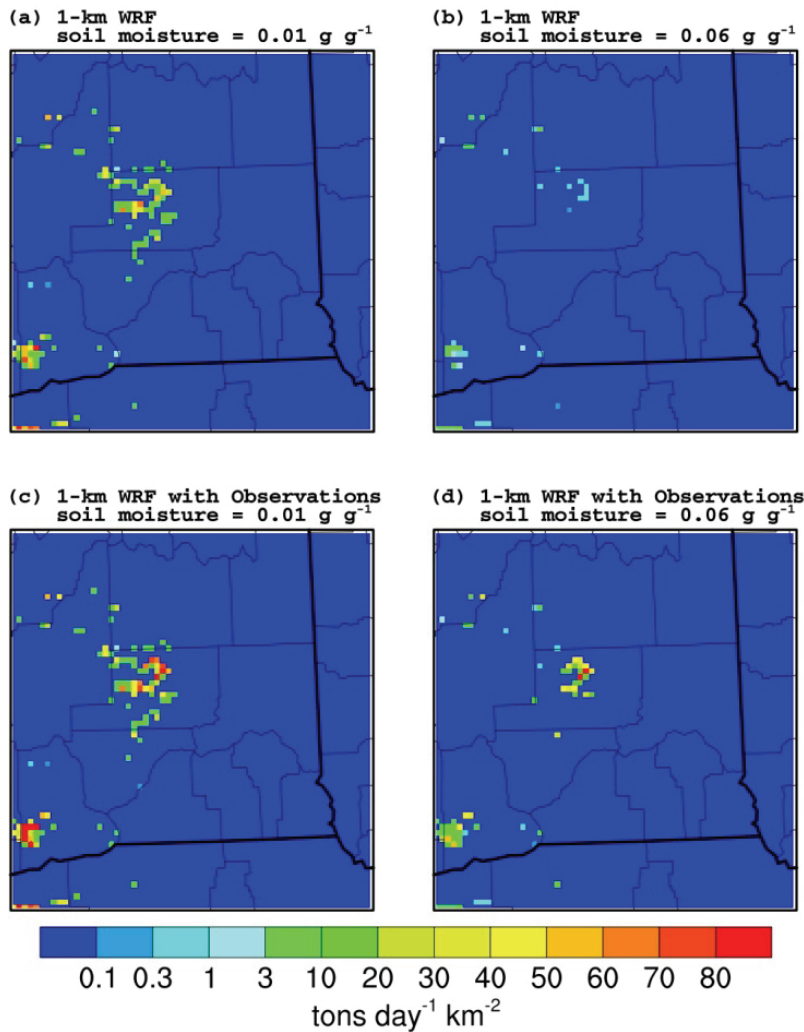
Figure 8. Spatial distribution of predicted 24 h PM_{10} emissions over domain 3 for 26 August 2010 based on 1 km WRF and EROSION simulations. The upper panels show results based on unmodified WRF wind speeds, and the lower panels show results when WRF wind speeds were adjusted according to observations. The left panels are based on surface soil moisture of 0.01 g g^{-1}, and the right panels are based on surface soil moisture of 0.06 g g^{-1}. One-kilometer emission results have been aggregated to 4 km cells to aid visualization.

tions in domain 3 when the surface moisture level is set to 0.01 and 0.06 g g^{-1}, respectively. Compared to the EROSION simulations without adjusting WRF wind speeds, the PM_{10} emission rates increase by a factor of 2 to 10 depending on the assumption of surface soil moisture. For surface soil moisture at the high end (to 0.06 g g^{-1}), increasing wind speeds leads to a large (factor of 10) increase in domain-total PM_{10} emissions because the number of grid cells reaching the threshold friction velocity more than doubles, as seen by comparing figures 8b and 8d. However, when the surface soil moisture is assumed to be only 0.01 g g^{-1}, correcting for the wind speeds resulted in only a two-fold increase in domain-total PM_{10} emission rates. Figures 8a and 8c indicate that the reason for this smaller relative increase is that the predicted amount of surface area undergoing erosion stays the same when wind speeds are adjusted higher at drier surface soil conditions.

PM_{10} Concentrations

PM_{10} emission rates generated with the EROSION submodel were used as input to the CMAQ chemical transport model to determine the air quality impact of windblown dust. Figure 9 shows the spatial distribution of the 24 h average PM_{10} concentrations for three of the 28 cases summarized in figure 7. Here we focus on results based on 1 km EROSION simulations because the results discussed above indicate that 4 km and 12 km EROSION simulations did not predict sufficient erosion. We also focus on the cases with surface soil moisture of 0.01 g g^{-1} because such dry surface soil conditions lead to regional dust storms that cause greater environmental impact than smaller events. As expected from the comparison of observed and modeled wind fields (figs. 5 and 6), figures 9a and 9b indicate that the spatial extents of the dust plume within domain 2 are similar for the 12 km and 4 km WRF/CMAQ cases. For the 12 km and 4 km WRF/CMAQ cases, the model predicts that 18% and 29%, respectively, of the area in domain 2 exceed the NAAQS of 150 μg m^{-3} for 24 h PM_{10} average. However, one advantage of using 12 km WRF/CMAQ is that, for the same amount of computing time, it allows for a larger domain to be simulated than does the 4 km WRF/CMAQ, and thus the former can be used to evaluate
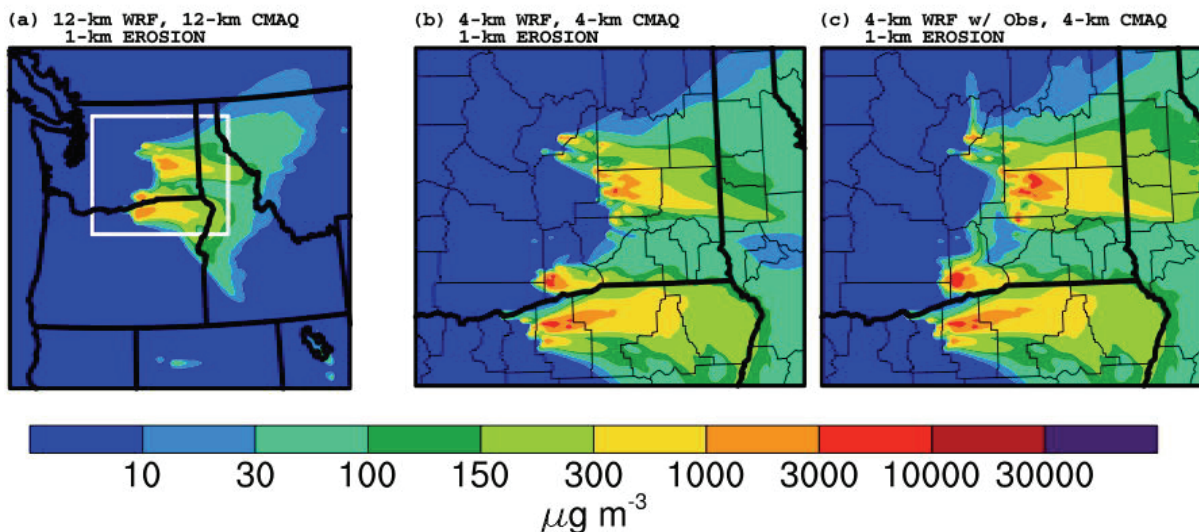
Figure 9. Spatial distribution of predicted 24 h average PM$_{10}$ concentrations for 26 August 2010 based on (a) 12 km WRF and CMAQ, (b) 4 km WRF and CMAQ, and (c) the same as (b) except that EROSION was run with WRF wind speeds adjusted according to observed wind speeds. All results shown are based on 1 km EROSION simulations. In (a), the white rectangular box indicates domain 2.

the impact of windblown dust much farther downwind of the source region. For example, figure 10a shows that the dust plume during the 26 August 2010 event reached as far as southwestern Idaho, a result consistent with the elevated PM$_{10}$ concentrations (peak of ~250 µg m^{-3}) measured at Nampa, Idaho, on that date (http://airquality.deq.idaho.gov; data not shown). Figure 10c shows the spatial distribution of 24 h PM$_{10}$ concentrations for the case in which the EROSION simulation used wind speeds that were adjusted according to observations. Compared to using 4 km WRF wind speeds without adjustment, adjusting the wind speeds increased the area in domain 2 that exceeds the NAAQS from 18% to 23%.

Figure 10 shows a comparison between modeled and observed hourly PM$_{10}$ concentrations for six sites in the Columbia Plateau region. PM$_{10}$ concentrations at Pullman were measured using an Aerodynamic Particle Sizer (APS spectrometer 3321, TSI, Inc., Shoreview, Minn.). The particle number size distribution measured by the APS was integrated for diameters between 0.5 and 10.0 µm and then converted to mass using an assumed density of 2.5 g cm^{-3}. PM$_{10}$ concentrations at the other five sites were measured using tapered element oscillating microbalances (TEOMs). PM$_{10}$ data for Burbank, Cheney, and Kennewick are from the Washington Department of Ecology (https://fortress.wa.gov/ecy/enviwa/Default.htm); PM$_{10}$ data for Sandpoint and Pinehurst are from the Idaho Department of Environmental Quality (http://airquality.deq.idaho.gov); and PM$_{10}$ data for Pullman were measured by the authors on the WSU campus. Results shown are based on 4 km WRF/CMAQ and 1 km EROSION simulations for domain 2; results from 12 km and 1 km WRF/CMAQ are similar and thus are not shown. Overall, model results agree best with the observations when the surface soil moisture was assumed to be between 0.01 and 0.03 g g^{-1}. When the WRF wind speeds were adjusted based on observations to drive the EROSION model, predicted PM$_{10}$ concentrations increased by a factor of 2. The model performed best at Cheney, Kennewick, and

Burbank. Although the model predicted the dust plume to travel to Sandpoint in northern Idaho, the model did not approach the observed peak of ~800 µg m^{-3} at that site. A likely reason is that the modeled wind direction was more westerly and less southwesterly than the actual wind fields. This is also consistent with the model overpredicting PM$_{10}$ concentrations in Pullman; if the modeled winds were more southerly, less of the dust plume would reach Pullman and more of the plume would reach Cheney and Sandpoint. However, without more spatially distributed ambient measurements, we cannot rule out the possibility that erosion occurred at some locations upwind of Sandpoint that were not predicted by the model.

THE 4 OCTOBER 2009 EVENT

The 4 October 2009 dust storm was captured by the MODIS sensors on NASA's Aqua and Terra satellites. These images provided a unique opportunity to qualitatively evaluate how the WRF-EROSION-CMAQ modeling framework captured the source locations of windblown dust and the spatial coverage of this event. Figure 11 shows the predicted surface-level PM$_{10}$ concentration from 4 km WRF/CMAQ and 1 km EROSION simulations assuming surface soil moisture of 0.01 g g^{-1} at 11:00 a.m. PST. These results are overlayed onto the MODIS image from the Terra satellite, which had an overpass time of shortly after 11:00 a.m. PST. An image captured by MODIS from the Aqua satellite approximately 3 h later and modeled results at 2:00 p.m. PST both indicate the storm was still active at that time with very similar spatial extent (figure not shown). Comparison of the MODIS image with measured PM$_{10}$ concentrations at the Burbank and Kennewick sites suggests that the MODIS sensors appear to detect the dust plume only when surface PM$_{10}$ concentrations are greater than ~300 µg m^{-3}; thus, cells in which modeled PM$_{10}$ concentrations are less than 300 µg m^{-3} are not shown in figure 11. The general trends in modeling results between various cell sizes and assumptions on surface soil moisture
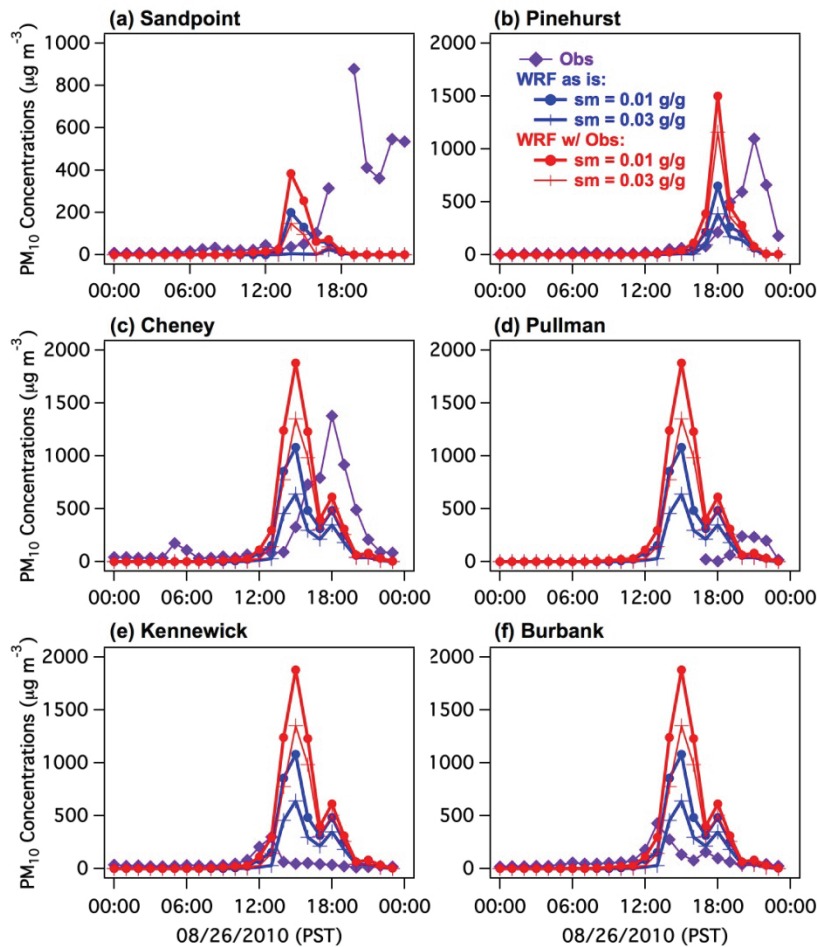
Figure 10. Observed and modeled hourly surface-level PM$_{10}$ concentrations at six sites for 26 August 2010; locations of the sites are shown in figure 4. All modeled results shown are based on 4 km WRF/CMAQ and 1 km EROSION simulations for domain 2 (sm = soil moisture).

were very similar to those for the 26 August 2010 case and are not repeated here.

The MODIS image (fig. 1 and background in fig. 11) indicates that the most significant erosion occurred in Lincoln, Adam, and Grant counties. The model also predicted a large source of windblown PM$_{10}$ emissions in Adams and Grant counties, consistent with the MODIS images. While the model did predict erosion in Lincoln County, the predicted PM$_{10}$ emission rates were too small to have a detectable impact on modeled PM$_{10}$ concentrations. Comparing the measured and modeled wind speeds at the Davenport site in northeastern Lincoln County indicates that the model underpredicted wind speeds by ~6 m s^{-1} (data not shown); thus, underprediction of wind speed in central Lincoln County is a likely cause of the underprediction of PM$_{10}$ emissions there. The model predicted erosion in Douglas County, but no such erosion was visible in the MODIS images. It is possible that erosion occurred, but the dust plume was too small to be detected from space. Modeled wind speeds at St. Andrews in Douglas County agreed with measured wind speeds, indicating that the model did not overpredict PM$_{10}$ emissions due to overprediction of wind speeds. Possible reasons for the overprediction by the model include the following: (1) the soil is not bare in contrast with the model assumptions, and (2) the assumed surface

soil moisture of 0.01 to 0.06 g g^{-1} was too low. In addition to the major source regions discussed above, the model also predicted lower rates of PM$_{10}$ emissions at a few other locations throughout the Columbia Plateau region. These smaller incidents are either too limited to be seen from space or their locations were under clouds or the dust plume and thus were not detected.

Figure 11 shows that the dust plume traveled southwestward, extending into northern Oregon. Because of heavy cloud cover, the MODIS sensors were unable to detect how far south into Oregon the plume traveled. The model also predicted the dust plume to travel southwestward; however, the modeled plume was more easterly and less northerly than the actual plume. Figure 12 shows wind rose plots based on hourly wind fields for 74 AgWeatherNet sites (http://weather.wsu.edu) and corresponding 4 km WRF cells. It indicates that the modeled wind directions were indeed less northerly than the observations.

Because of the source region of the dust and the wind directions, the dust plume did not travel past Cheney and Pullman in eastern Washington, and only the Burbank and Kennewick PM$_{10}$ monitoring sites detected elevated PM$_{10}$ concentrations on 4 October 2009. Figure 13 shows the observed PM$_{10}$ concentrations at these two sites, which are located only ~17 km apart and just at the edge of the dust
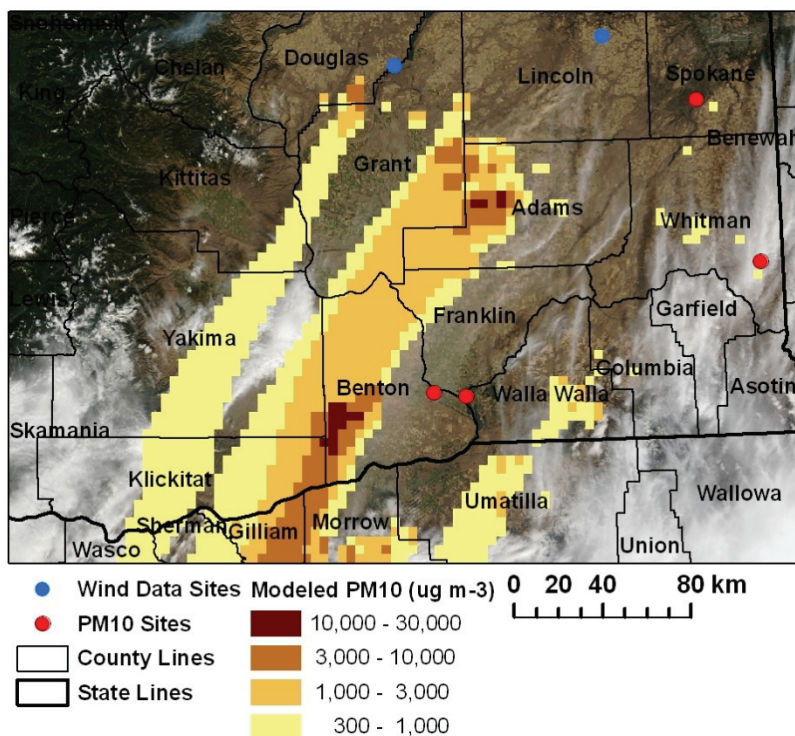
Figure 11. Predicted surface-level PM$_{10}$ concentrations from 4 km WRF/CMAQ and 1 km EROSION simulations assuming surface soil moisture of 0.01 g g^{-1} at 11:00 a.m. PST overlaying MODIS image from the Terra satellite (overpass time shortly after 11:00 a.m. PST) (see fig. 1 for credits). For clarity, cells with modeled PM$_{10}$ concentrations less than 300 μg m^{-3} are not shown. Names of the wind and PM$_{10}$ measurement sites are given in figure 4.
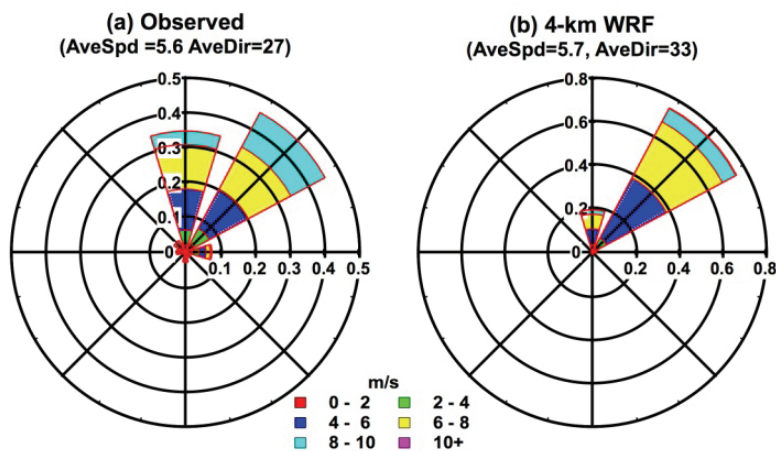

Figure 12. Wind rose plots based on hourly wind fields for 74 AgWeatherNet sites (http://weather.wsu.edu) and corresponding WRF cells in domain 3 for 4 October 2009: (a) observation and (b) 4 km WRF.

plume. While the timing of peak PM$_{10}$ concentrations was very close at these two locations, the observed peak PM$_{10}$ concentrations at Kennewick were higher by a factor of 2 to 3 than those at Burbank, indicating that the PM$_{10}$ loading decreased rapidly away from the centerline of the plume. Because the modeled dust plume traveled more westward than the actual plume, the modeled plume missed Kennewick and Burbank. Figure 13 includes predicted hourly PM$_{10}$ concentrations near the edge of the modeled plume that is four cells westward (~16 km) of the cell in which Kennewick is located. Even though the model missed the direction of the plume slightly, it captured the timing of the peak PM$_{10}$ concentrations. This confirms the

capability of the new WRF-EROSION-CMAQ regional air quality modeling framework to predict the onset of wind-blown dust storms and qualitatively capture the spatial extent of such storms in the Columbia Plateau region.

MODEL UNCERTAINTIES AND FUTURE NEEDS

With the aid of satellite data, we have shown that the WRF-EROSION-CMAQ regional modeling system was capable of simulating two recent dust storms in the Columbia Plateau region of Washington and Oregon. Comparisons of modeled results with available observational PM$_{10}$ concentration data from six sites in the region and with a satellite image of the 4 October 2009 event suggests that
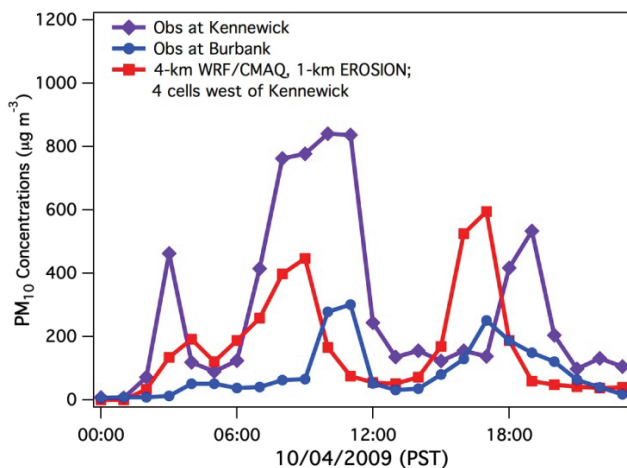
Figure 13. Hourly PM$_{10}$ concentrations observed at Kennewick and Burbank and predicted at the cell located four cells (~16 km) to the west at the cell in which Kennewick is located. Observed data are from the Washington Department of Ecology (https://fortress.wa.gov/ecy/enviwa/Default.htm).

the modeling system performed well in capturing these two windstorm events in terms of timing and spatial coverage of the PM$_{10}$ plume. However, a more rigorous and quantitative evaluation cannot be readily performed because various assumptions were made regarding the surface conditions (e.g., surface soil moisture) for which very little information is available, and because there is a lack of PM$_{10}$ emission and finer-resolution PM$_{10}$ concentration data. Even though a more rigorous evaluation of the model is not currently possible, this study points toward several areas of focus for future model improvement.

A major source of uncertainty in the model results is the underprediction of peak wind speeds associated with windstorms. This underprediction occurs regardless of the size of grid cells used. Under very dry surface soil conditions, the error associated with underprediction of wind speeds is about a factor of 2 in PM$_{10}$ emission rates. The relative error is even greater when surface soil moisture is higher, although error under such conditions is less critical because higher surface soil moisture is associated with smaller dust storms for which environmental impact is less than that of larger storms. For retrospective analysis, model wind speeds can be adjusted using observational data, as was done in this study. However, for forecasting purposes, observational data will not be available. Thus, it is critical to improve model wind speed predictions for more accurate predictions of PM$_{10}$ emission rates and concentrations in forecasting applications.

Another major source of uncertainty in the model results is the surface soil moisture, which affects the predicted PM$_{10}$ emission rates by a factor of 10. The current modeling framework requires surface soil moisture to be an input to the model, but there is no observational data available at the spatial and temporal resolutions required to predict erosion. In the full implementation of the WEPS model (Gao et al., 2013), soil moisture is predicted at spatial scales of 1 km across the simulation domain. It will be important to perform field measurements at various locations to evaluate how well the model predicts surface soil moisture and

quantify the model error in PM$_{10}$ associated with that of surface soil moisture.

A source of uncertainty in the EROSION results is the errors associated with the aggregation of soil properties. We used the soil database of Feng et al. (2009), which is based on the STATSGO database. Feng et al. (2009) took the soil components within a single map unit and aggregated the soil properties to create a single soil component. This aggregation of soil properties likely introduced error in the EROSION results, especially if there is large variability in the soil component properties for a map unit. The nonlinear response to the soil properties on erosion susceptibility means that using aggregated properties as inputs to EROSION does not necessarily give the same results obtained by area-weighting EROSION simulations for each soil component individually. Additional simulations are required to address model errors introduced by the aggregation of soil properties applied in this study.

Even though WEPS is more process-based in comparison to previous windblown PM$_{10}$ emission models that have been applied for regional air quality modeling, WEPS still contains empirical equations and parameters based on measurements of soils from a finite number of locations. The need to modify the static bare-soil threshold friction velocity (u_{*tb}) to simulate erosion events in the Columbia Plateau region highlights this fact. The modification was necessary because the original value of u_{*tb} calculated by WEPS (~0.5 m s^{-1}) was too high in comparison to the measured value of 0.3 m s^{-1} or less (Feng and Sharratt, 2007b; Sharratt and Vaddella, 2012). One possible reason is that the estimated soil aggregate size distributions (ASD) are inaccurate for characterizing the fine, silt loam soil of the Columbia Plateau region. We used the ASD estimates of Feng et al. (2009), who based their calculations on the sand, silt, and clay fractions of the soil components in the STATSGO database. The estimated ASD may be too high, which would lead to overestimation of u_{*tb}. Another potential limitation of WEPS is that it uses the same threshold friction velocity for emissions by both direct suspension and saltation processes, whereas field observations indicate that, for the study region, emissions can occur by direct suspension process when saltation is not a major mechanism for eroding soil or generating dust emissions (Kjelgaard et al., 2004). Further field and laboratory work is necessary to develop a robust relationship between surface soil properties and separate threshold friction velocities for PM$_{10}$ emissions from direct suspension versus saltation.

CONCLUSION

The EROSION submodel of the Wind Erosion Prediction System (WEPS) has been modified from a single-field model to be run on multiple cells independently. It was then incorporated into a regional air quality modeling framework for the Pacific Northwest. The capability of the WRF-EROSION-CMAQ modeling system with the aid of satellite data was demonstrated by retrospective simulations of two recent dust storms in the Columbia Plateau region of Washington and Oregon. Model results were compared

against observed wind fields and PM_{10} concentrations as well as satellite images of the 4 October 2009 event. The model results were relatively insensitive to the size of the WRF and CMAQ grid cells typically used for weather and air quality forecasting studies. However, it is clear that even if relatively large cells must be used for the meteorological and transport modeling due to computational constraints, there is still great benefit in using MODIS NDVI data at a finer scale to identify cells that have erodible surfaces and running EROSION at 1 km scale. The WRF model systematically underestimated the peak wind speeds during the dust events, resulting in PM_{10} emission rates that were lower by a factor of ~2 during very dry surface soil conditions. The model results were also very sensitive to the assumption of surface soil moisture. This underscores the need to incorporate the HYDROLOGY submodel to predict surface soil moisture, as no data are available at the spatial and temporal resolutions required. Directly applying the AMSR-E soil moisture data led to underprediction of PM_{10} concentrations because AMSR-E detects the moisture level for a deeper soil column than that which governs the erosion processes.

Evaluating how well the model can simulate the source locations of windblown dust and the spatial extent of the dust plumes is difficult due to the low density of ambient PM_{10} monitors in the region. For the two storms simulated in this study, the new WRF-EROSION-CMAQ modeling framework performed well in predicting the onset and the timing of dust storms. Within the region for which the MODIS sensors can detect the dust plume (i.e., no clouds), the model appears to predict well the spatial extent of the dust plumes for the 4 October 2009 event. The model performance suggests that the new framework has potential to be used as a warning system for dangerous road conditions. However, model improvements as well as further statistical evaluations are needed before the modeling framework can be used for quantitative assessment of the air quality impact of windblown dust.

Future work will incorporate the full WEPS model into the regional air quality modeling framework for the whole Pacific Northwest region. Currently, the full WEPS model has only been incorporated for the state of Washington (Gao et al., 2013). Results presented here indicate that it will be critical to evaluate how the full WEPS model can simulate surface soil moisture. Refinement also must be made to improve how modeled wind speeds from WRF are to be used by WEPS because the WRF model has a tendency to underestimate high wind speeds during the low-pressure events that trigger wind erosion in the region. Also important is refinement of the EROSION submodel to predict threshold friction velocity based on the soil aggregate properties and to model emissions from direct suspension and saltation independently. Targeted field measurements combining surface conditions and erosion rates at the source with atmospheric PM_{10} concentrations at multiple locations downwind will provide the most value in evaluating the new modeling framework.

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