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Large offshore wind farms have minimal direct impacts on air quality

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Abstract

Wind power has rapidly grown over the past decade because it is clean, renewable, and abundant. However, wind farms can affect local weather conditions and possibly alter the transport, diffusion, and concentration of air pollutants. Given the unprecedented expansion of offshore wind farms planned along the U.S. East Coast by the Bureau of Ocean Energy Management (BOEM), This study aims to investigate if and how those future wind farms might directly affect air pollution along the U.S. East Coast, in particular the levels of ozone (O_3), fine particulate matter ($PM_{2.5}$), sulfur dioxide (SO_2), and nitrogen dioxide (NO_2). These pollutants are regulated at the federal and state levels and are harmful to human health. We exclusively study the direct effects of the wind turbines on air pollution (via meteorological changes), rather than investigating the indirect impacts of replacing fossil-fuel power plants with wind farms. We first run a numerical meteorological model, the Weather Research and Forecast (WRF) model, to simulate the meteorology along the U.S. East Coast during the summer of 2018 in two scenarios, with and without the wind farms. Then we use the output of these two sets of simulations from the WRF model as input to the Comprehensive Air Quality Model with extensions to simulate the changes in air quality in the study domain due to the wind farms. On average, we only find a minor increase in O_3 levels within the wake of the New Jersey WEA. The minor changes to O_3 can be attributed to the slight temperature increase below the turbine hub height, within the rotor area, as well as a significant decrease in wind velocity in the wake of the turbines and a slight increase in volatile organic compound levels. In addition, we report that the other three pollutants remain unchanged in the presence of wind farms. In summary, the direct impacts on air pollution by the BOEM-planned offshore wind farms are expected to be negligible.

1. Introduction

Wind is one of the most abundant energy sources in nature, making it an excellent resource for generating clean and renewable energy for a sustainable world. The benefits of wind energy in mitigating air pollution and greenhouse gas emissions while generating clean and renewable electricity are undeniable. However, wind turbines have the potential to alter the surrounding atmosphere through the creation of wakes. Wakes are formed as wind turbines extract kinetic energy from the air, resulting in regions characterized by lower wind speeds and higher turbulent kinetic energy (TKE) compared to the undisturbed wind upstream [4, 47]. Numerous studies have demonstrated that wind farm wakes can reduce wind speeds by 2–2.5 m s^{-1} [1, 20, 48] and travel distances exceeding 20 km [43, 48], with some wakes extending even further, reaching distances of 40–70 km [1, 18, 22] and up to 150 km [20], depending on factors like atmospheric stability or topography [48]. This suggests that the wakes of offshore wind farms might impact surface temperature and other atmospheric properties not only offshore

but potentially also onshore along the coast if the wakes are long enough to reach the land. Thus, large wind farms can influence meteorology and air quality through two primary mechanisms. First, they reduce anthropogenic emissions and air pollutants by replacing fossil fuels, which we refer to as *indirect impacts* [8, 31, 38]. Second, they affect local and downstream meteorological conditions by extracting kinetic energy from air flows that can impact the pollution levels, which we refer to as *direct impacts* [18, 20, 24, 28]. While extensive literature has explored the indirect benefits of wind power, consistently demonstrating significant reductions in air pollutant emissions, in this study, we only focus on the direct impacts of the large wind turbine clusters on air pollution concentrations.

Air pollutant concentrations are strongly correlated with meteorological variables such as wind speed, temperature, humidity, and atmospheric stability. For instance, in an unstable atmosphere, because of strong mixing, the depth of the boundary layer and therefore the height of the inversion layer that caps the unstable layer is higher [5, 23, 42, 45]. In a stable atmosphere, by contrast, the heat fluxes are downward, therefore the air does not tend to rise but it tends to sink [49, 3]; the same for the pollutants in the air. Depending on atmospheric stability, pollutants can travel long distances before hitting the ground, or they can get stuck under the capping inversion layer and hit the ground quickly [5, 23, 42]. Another essential factor in pollution transport is wind speed. Stronger winds can transport air pollutants longer distances, whereas weak winds help the pollutants stay near their emission source, which results in higher pollution levels near the source and low-range transport of pollution [37]. According to [33], in the northeastern U.S., higher wind speeds often impact the transport of ozone precursors and consequently result in higher ozone levels. Surface temperature is another important player in pollution levels in a region. For instance, ozone pollution is strongly correlated with temperature, as ozone concentrations increase with higher temperatures and in the presence of sunlight [42].

Here we focus specifically on four pollutants: fine particulate matter, nitrogen dioxide, sulfur dioxide, and ozone. These four pollutants are called 'criteria' pollutants because they are regulated at the federal level by the U.S. Environmental Protection Agency (EPA) via the National Ambient Air Quality Standards [16, NAAQS;]. Fine particulate matter ($PM_{2.5}$) is a harmful air pollutant that consists of microscopic solid or liquid particles with a diameter smaller than 2.5 μm. These particles can penetrate human lungs and even the bloodstream and cause serious health problems [12]. Nitrogen dioxide (NO_2) is harmful to humans by irritating the human respiratory system [13] and to the environment by creating acid rain [29]; it is also a precursor to tropospheric ozone (hereafter referred to as ozone). Ozone (O_3) is a secondary pollutant, meaning that it is not directly emitted into the atmosphere but is instead formed from chemical reactions between nitrogen oxides (NOx) and volatile organic compounds (VOCs) in the presence of sunlight [25]. Exposure to O_3 can irritate the eyes, nose, and throat. It can also worsen existing lung conditions and pose a higher risk of premature death, particularly for individuals with heart or lung diseases [14]. Similarly, short-term exposure to sulfur dioxide (SO_2) can harm the human respiratory system.

Here, we investigate the extent to which the wakes of large offshore wind farms potentially interfere with the transport and chemistry of air pollutants in coastal areas along the U.S. East Coast. The federal agency in charge of planning, permitting, and leasing offshore wind energy areas (WEAs) in the U.S. is the Bureau of Ocean Energy Management (BOEM). BOEM has sold approximately 42 GW of offshore wind leases for over $5.44 billion and over 2 GW are approved or under construction as of 2024 [7]. Because of this unprecedented growth of the number of turbines in the waters, concerns are rising about possible meteorological and air quality impacts offshore and possibly inland. For example, Mo *et al* [32] revealed an 'edge effect,' which is slightly higher NO_2 levels at the immediate upwind and border region of the wind farm and lower NO_2 concentrations within the wind farm. However, their analysis was based on simulations spanning only 5 d in November over a wind farm in China, whereas our study extends the simulation period to three months over the summertime when O_3 is at its highest. In a similar study, Ruan *et al* [41] took a modeling approach using the WRF-CMAQ modeling system to study the direct and indirect impacts of wind turbines on air pollution levels. They reported a monthly net increase of 0.067 $\mu g\,m^{-3}$ (0.08%) relative to the regional average which is in agreement with our findings. On the other hand, Li *et al* [27] modeled the impact of wind farms using the WRF-Chem model and concluded that large-scale wind farms decrease $PM_{2.5}$ levels within the farms, while significantly increasing $PM_{2.5}$ concentrations by 49% in regions hundreds of kilometers downstream. They also report a range of −40%–250% change in $PM_{2.5}$ levels in different hours and interpret these changes as an impact caused by the wind turbines, directly. It is important to note that such significant changes in $PM_{2.5}$ concentrations are unusually high values for impacts induced by wind turbines, alone. In our previous work with the WRF model [20], we identified that the wind farm parameterization (WFP) in WRF can induce perturbations at the initial time, which, if not properly addressed, can lead to non-linear growth of model noise and create unrealistic results in regions

far from the wind turbines. The numerical noise is also discussed in the literature [2, 26]. It appears that the extreme variations in $PM_{2.5}$ concentrations suggested in that study, are likely artifacts of the model and have been misinterpreted as physical phenomena. We addressed these issues by employing techniques such as nudging in the outer domain and above the planetary boundary layer (PBL) to mitigate the artifacts. Additionally, we re-initiated our meteorology model every 54 h with a 6 h spin-up time to prevent the model from diverging from the solution, further reducing the potential for unrealistic results. Additionally, we limited our analysis to pollutant averages. By taking these precautions, we minimized the possibility of unrealistic effects that may have otherwise been misinterpreted as actual wind farm-induced changes in pollution levels.

Temperature changes induced by wind farms appear to be strongest during stable atmospheric conditions and stable conditions are dominant during the summer in the marine environment [3, 20]. In addition, ozone pollution is highly correlated with temperature, and the highest ozone episodes are found during the summer months. For these two reasons, we base our study on a summer (i.e. three months June through August), in particular the summer of 2018, which is the year with the most recent emission data from the Long Island Sound Tropospheric Ozone Study [34, LISTOS;] lead by the Northeast States for Coordinated Air Use Management [36]. LISTOS is a multi-agency collaborative field campaign based on the existing observational networks in the area that provides higher resolution emission data for different purposes [46].

In our recent work [20], we found that building the BOEM-planned offshore WEAs along the East Coast of the U.S. will create wakes that, at the hub height of the wind turbines that can travel up to 150 km along the direction of the prevailing winds (southwesterly). These turbine wakes reach the surface within the areas of the offshore wind farms and cause a slight wind speed deficit at the sea surface there, but not inland. The additional TKE in the wake, which is created by the rotation of the blades in the near-wake region and by wind shear production in the far-wake region, increases the downward heat fluxes within the wake during stable atmospheric conditions, which are most common in the summer. This heat flux convergence creates warming below the hub height and cooling above the rotor. The opposite occurs during unstable conditions, i.e. slight cooling below and warming above the rotor. However, to understand if these changes may affect air quality in the coastal areas, it is necessary to simulate not only the air pollutant transport, diffusion, deposition, and removal processes, but also the chemical reactions that the air pollutants are subject to. For this task, it is necessary to use an atmospheric chemistry model capable of resolving all such physical and chemical processes. As described in the next section, we will use the output of the meteorological simulations from our previous work [20] as input to the Comprehensive Air Quality Model with extensions (CAMx) model.

2. Methods

2.1. Setup of the WRF-CAMx modeling system

The models used in this study are the WRF model version 4.3 and the CAMx model version 7.1 with the Carbon Bond version 6, revision 5 chemical mechanism. WRF was developed by the National Center for Atmospheric Research (NCAR) and is one of the most widely-used numerical weather prediction models [44]. The WRF output fields provide the meteorological inputs to CAMx. CAMx is a modular, Eulerian, 3-dimensional photochemical air quality model that simulates the emission, production, advection, diffusion, chemical transformation, and removal of atmospheric pollutants at regional scales [39] and is among the few that are recommended by the EPA for regulatory purposes [15].

We simulate the summer of 2018, the year of the most recent emission inventory by the NESCAUM, which covers 1 May 2018 to 1 October 2018 with a 4 km horizontal grid resolution, which is finer than that of the previous NEI emission inventories (12 km grid resolution). Since the 2018 emission files cover 315×300 grid cells horizontally and 35 layers vertically, we chose the same size for the CAMx domain, which includes all the offshore WEAs planned along the U.S. East Coast (figure 1). For the WRF simulations, the domain is slightly larger with 400×400 horizontal grid points (table 1). The gridded emissions are merged, meaning that they are provided as one set of surface emissions that include all sectors (e.g. traffic, residential, etc). On the other hand, the elevated point sources are provided for each sector separately, including the shipping emissions over the Atlantic Ocean.

We setup both WRF and CAMx models with a 4 km horizontal resolution, the same as that of the LISTOS emission data, to avoid spatial interpolation of the gridded emissions data. To minimize the impacts of the initial conditions on meteorology modeling results with the WRF model, we used a 6 h spin-up time for each 54 h WRF run to maintain model stability with the WFP. Next, we run CAMx continuously for each of the three months, using a restart mechanism and a 48 h spin-up time for each initialization to allow chemical processes to stabilize. In summary, WRF had a 6 h spin-up every 54 h, while CAMx had a 48 h spin-up each month (table 1).

To address the research question, we first simulate two different meteorological scenarios with WRF: the first is the control case, in which the simulations are

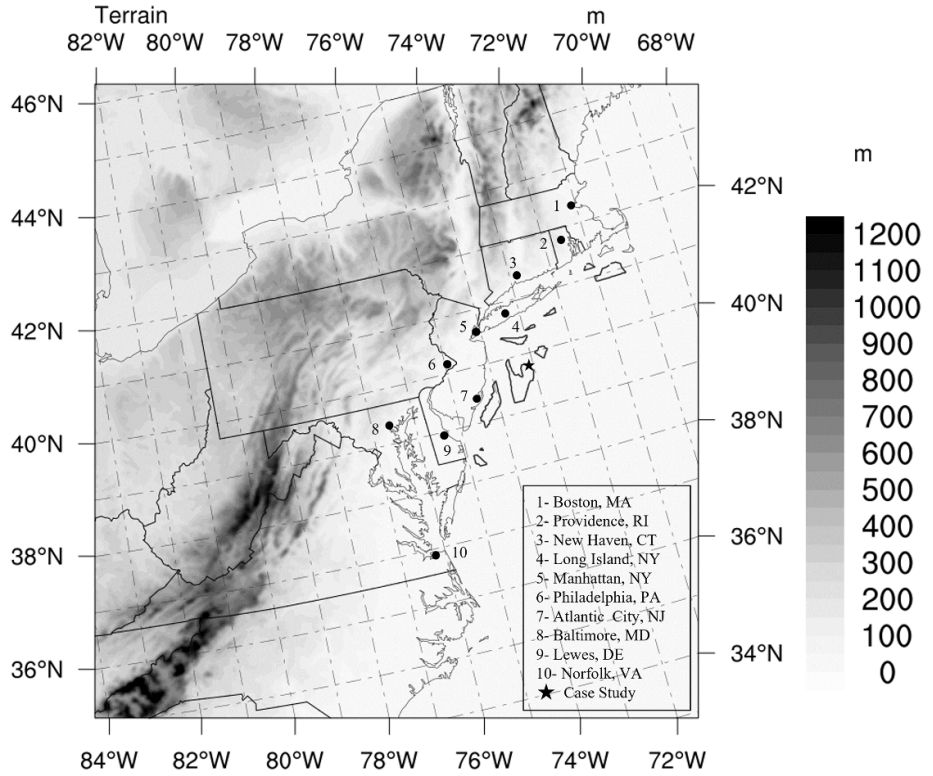

Figure 1. The CAMx domain with 315 × 300 grid cells and 4 km horizontal grid resolution. The numbered circles are the locations that have been used in the study. The WEAs are shown offshore in the Atlantic Ocean with black-line shapes. The star is the location of the case study within the New Jersey WEA.

Table 1. Configuration details of the WRF-CAMx model.

Simulation period	1 June 1–30 August 2018
Spin-up time	48 h
Horizontal grid resolution	4 km
Vertical layers	35
WRF model	Version 4.3 [44]
Wind farm parameterization	Fitch with TKE advection
wind turbine model	DTU-10MW
Wind turbine hub height	119 m
Wind turbine diameter	178 m
Initial/Boundary conditions	NAM reanalysis (6 h, 12 km resolution)
Land surface model	Noah-modified 21-category, IGBP-MODIS
PBL scheme	MYNN2
Shortwave radiation	RRTMG
Longwave radiation	RRTMG
SST update	NASA-JPL (1 km resolution) [50]
Grid size	400 × 400 grid cells
CAMx model	Version 7.1 [39]
Chemistry mechanism	Carbon Bond 6 (revision 5)
Meteorological inputs	WRF model (version 4.3)
Emission data	LISTOS campaign 2018 [36]
CMV emissions	Inline point sources
Initial/Boundary conditions	EPA 2018
Grid size	315 × 300 grid cells

carried out without any wind farms, and the second includes all the wind farms planned by BOEM in the WEAs ('WithFarms'). Details about the location of the leased and planned wind farm locations, as well as their power production, are obtained from [6]. We turn on the Fitch WFP in the WithFarms simulations to account for the extraction of kinetic energy and addition of TKE by the wind farms [18]. We use the North American Meteorology (NAM) model output with a horizontal resolution of 12 km [35] to provide the initial and boundary conditions to the WRF model for both scenarios every 6 h (table 1). The simulations cover the period from 00:00 UTC on 01 June 2018, to 00:00 UTC on 1 September 2018. The WRF runs are re-initialized every two days with the NAM to avoid potential numerical errors, following [20]. Every WRF run starts at 18:00 UTC on the prior day and continues for 54 h. The first 6 h of every simulation are considered spin-up time and are not included in any later analysis. The details of the WRF simulations are discussed thoroughly in [20].

In our previous work [20], we analyzed the difference between these two scenarios from a meteorological perspective to assess how wind farms along the U.S. East Coast could impact wind speed, temperature, humidity, and heat fluxes at the surface. In brief, wind speed decreased significantly (up to 3 m s^{-1} on average) at hub height due to the WEAs, while the wakes reached the surface and decreased the wind speed by 0.5 m s^{-1} on average at the surface. Surface temperature in the summer decreased above the rotor area, increased below hub height within the rotor area, and remained unchanged at the surface. Here, to complete that work, we use the same two meteorological scenarios as inputs to CAMx to quantify the direct impacts of the wind farms on air quality as follows.

Next, we run the CAMx model for the same two scenarios, i.e. the control case with no wind farms and the WithFarms case with all the BOEM-planned WEAs operational. The difference between these two CAMx simulations gives a clear quantification of the direct impacts of wind farms on the air pollutant concentrations—specifically O_3, $PM_{2.5}$, NO_2, and SO_2—in the domain of interest, including in the coastal areas. We note that indirect impacts of the WEAs, such as the displacement of fossil-fuel burning power plants and their greenhouse gas emissions, are not accounted for here, only the direct impacts of the WEAs caused by the slightly altered meteorological conditions. The boundary and initial conditions for the CAMx simulations were also provided by the LISTOS campaign [34]. We run the CAMx model continuously for the same three months as the WRF runs (table 1). We use the initial conditions provided by the EPA for the first three simulation days only; for the following days, we use the restart mechanism, where the outputs of the run for the previous day provide the initial condition for the current day.

We note that for the validation in section 3, we only compare the simulation results from the control case against measured air pollutant concentrations, as only one small offshore wind project had been built by the year 2018, i.e. 30 MW at Block Island. For the air pollutant observations, we rely on measurement data sourced from the EPA's Air Quality System (AQS) [17].

3. CAMx model performance analysis

The primary goal of this study is to explore direct changes to air pollutant concentrations between the two scenarios, one with the turbines and the other without. Since any model bias and uncertainty remain the same in both scenarios and since we analyze the differences between the two scenarios, model bias, and uncertainty are unlikely to have a significant influence on our analysis because they are eliminated by the difference-in-differences approach. Nevertheless, here we present an evaluation of CAMx model performance for the control scenario, which is the only one that is comparable with observations in 2018.

For the ground measurements, we rely on observed concentrations sourced from EPA's AQS, which is publicly accessible [17]. It is worth mentioning that evaluating $PM_{2.5}$ is challenging because the EPA-reported $PM_{2.5}$ concentrations are directly obtained through instrument measurements, classifying any particle smaller than 2.5 micro-grams as a $PM_{2.5}$ species. This method does not distinguish between the various chemical constituents of the particles detected by the measuring instruments. By contrast, the simulated $PM_{2.5}$ species in our study are defined based on CAMx model documentation [39], which accounts for chemical speciation. In addition, the $PM_{2.5}$ mass measurement methods require that filters be equilibrated for 24 h at a constant relative humidity between 30% and 40% to minimize the liquid water content [11, LWC;]. As such, the measured $PM_{2.5}$ is closer to dry than moist particulate. In CAMx, LWC can be included or not in the calculation of $PM_{2.5}$ concentrations via the species called PH_2O. Unless otherwise stated, here we will not include LWC in the simulated $PM_{2.5}$ concentrations; however, the effect of LWC is evaluated in this section because it is potentially relevant given the high relative humidity conditions in the offshore environment.

To effectively compare simulated and measured concentrations, it is important to mention that the regulatory framework established by the EPA requires that each criteria pollutant be averaged in a different way and over different time periods to obtain a statistic called 'design value,' which is ultimately compared against the values in the NAAQS. The calculation of the design value is rather complicated. For example, a region violates the O_3 standard if

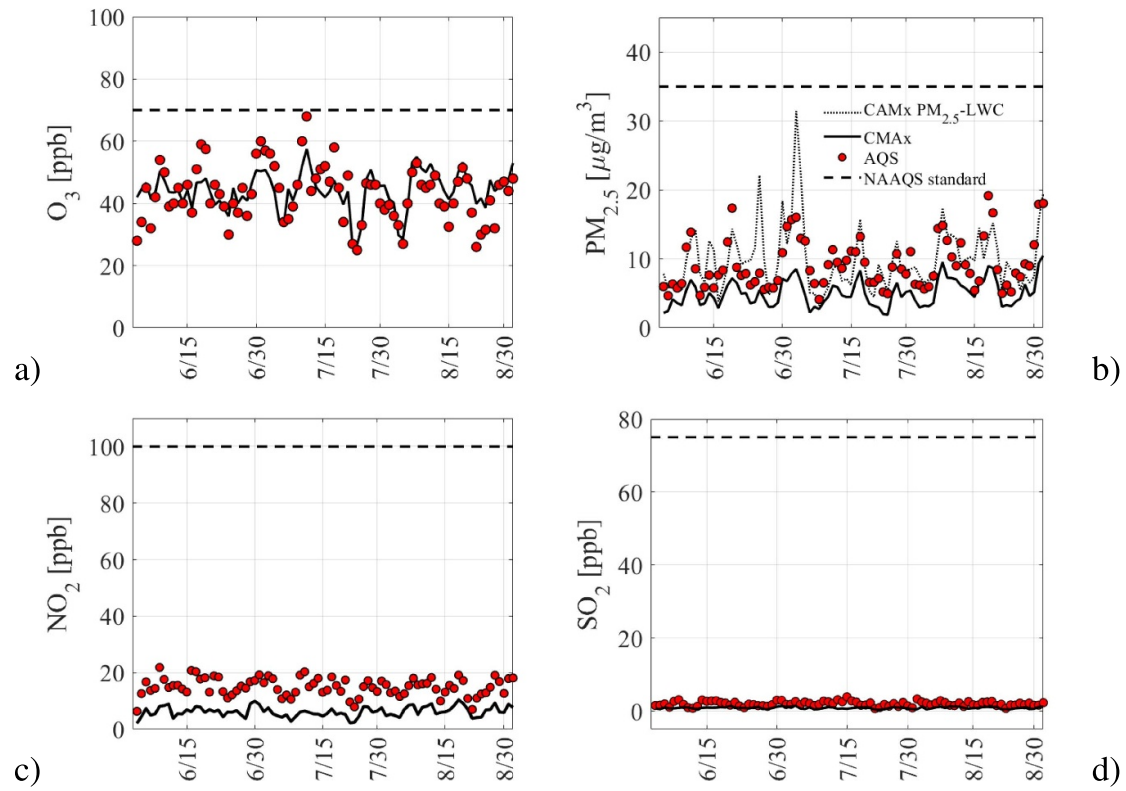

Figure 2. Time series of average concentrations simulated by CAMx versus measurements over the period of interest in summer 2018: (a) daily maximum of 8 h average O_3 (ppb), (b) 24 h average $PM_{2.5}$ ($\mu g\,m^{-3}$) with and without LWC, (c) daily maximum of 1 h average NO_2 (ppb), and (d) daily maximum of 1 h average SO_2 (ppb). The averages are calculated across all AQS stations (red dots) and all grid cells containing the AQS stations (black solid lines) for each day. This figure is reproduced from [19]. CC BY 4.0

the three-year average of the yearly 4th highest daily maximum 8 h average ozone concentrations exceeds 70 ppb [16]. In this study, it is not possible to calculate actual design values because we only simulated one summer (as the gridded emissions are provided for summer 2018 only), whereas three full years of simulations would be required. Thus, we could not calculate the design values for ozone at the locations of the monitoring sites in compliance with the NAAQS. Similarly for $PM_{2.5}$, NO_2, and SO_2, the NAAQS need to be compared against three-year metrics [16]. As such, in this section, we utilize the following statistics that are close to the design values but are obtained over the three-month study period of summer 2018:

- O_3: daily maximum 8 h average;
- $PM_{2.5}$: daily 24 h average;
- NO_2: daily maximum 1 h average;
- SO_2: daily maximum 1 h average.

For O_3, we use the forward average of hourly CAMx-simulated concentrations, as recommended by EPA [10, 17], to calculate the 8 h averages.

Looking at the time series of observed versus simulated concentrations at the locations with AQS stations (figure 2), it appears that CAMx performs satisfactorily in general. For O_3, CAMx captures the correct magnitude and trends of the concentrations, with a tendency to under-estimate peak concentrations during the day and over-estimate concentrations at night (figure 2(a)). The mean bias error (MBE) is -1.12 ppb, which indicates a systematic underestimation of around 2.5% across the monitoring stations within the designated domain.

For $PM_{2.5}$, the inclusion or exclusion of LWC from the calculation of the simulated concentrations impacts the model performance (figure 2(b)). Without LWC, CAMx systematically underestimates $PM_{2.5}$ concentrations, with an MBE of $-5\,\mu g\,m^{-3}$, as is commonly observed in prior studies [9, 21]. However, if LWC is accounted for (dashed line in figure 2(b)), the negative bias is reduced for most days, but at the expense of large over-estimates during a few high $PM_{2.5}$ episodes. As such, we do not include LWC in the calculation of simulated $PM_{2.5}$ concentrations in the rest of this study.

The model systematically underestimates NO_2 concentrations (figure 2(c)), which aligns with the literature [30]. An accepted explanation is that the monitoring stations are typically situated close to major roadways characterized by heavy traffic flow, resulting in an inherent positive bias in the observations. Lastly, CAMx showcases a strong alignment with observational data in terms of SO_2

simulations with minimal deviation from the observations (figure 2(d)).

We note that the observed and simulated concentrations of $PM_{2.5}$ (without LWC), NO_2, and SO_2 are well below the NAAQS values and therefore even a potentially large error by CAMx will not impact the attainment status of a monitoring site.

4. Results and discussion

In this section, we discuss our findings for changes in each pollutant. The goal of this study is not to determine the net changes in pollution changes through the replacement of fossil fuels by wind farms but to understand how wind farms impact local pollution levels by inducing changes in meteorological factors such as wind velocity, temperature, and humidity.

In section 4.1 we discuss the four pollutant average concentration changes calculated as the difference between the WithFarms results and those of the Control case (with no offshore wind farms). Next, in section 4.2 we focus on ozone only because it is the only pollutant among the four considered here whose concentration changes induced by the offshore WEAs can occasionally extend to the coastal areas.

4.1. Average concentrations

Starting with ozone, we find that, in general, the three-month average daytime O_3 concentrations are minimally affected by the presence of offshore wind farms (figure 3(a)). Along the coastlines and within a few tens of kilometers inland, localized decreases of up to 0.1 ppb are found. The only regions with a slight increase in ozone concentrations, up to 0.2 ppb, are at and downwind of the offshore wind farm locations over the Mid-Atlantic (figure 3(a)). This amount is less than 1% of the typical 8 h ozone concentrations in the region, thus it can be considered insignificant.

In addition to the three-month averages, we analyze in more detail the hourly O_3 concentrations in the NJ wind farm (the star in figure 1) with and without the wind farms (figure 4(a)). It is clear that the impact of the wind farms is minimal, causing the slope of the regression between WithFarms and control results to be slightly larger than 1 but with a negative intercept (figure 4(a)).

We hypothesize that the slight increase in O_3 concentrations near the largest offshore wind farms may be due to a combination of temperature and wind speed changes. According to our previous study [20], although the temperature at the ocean surface was not significantly affected by the presence of wind farms in summer, it increased by 0.5 °C on average in the rotor area below the hub height. In addition, low wind speeds are favorable to the accumulation of ozone precursors (VOCs and NOx) and the subsequent formation of ozone, whereas higher wind speeds tend to dilute or disperse pollutants. In the presence of the turbines, in our previous study [20] we also found that wind speed decreased significantly, by up to 0.5 and 3 $m\,s^{-1}$ on average at the surface and hub height, respectively. Here, looking at the output data at the offshore location marked with a star in figure 1, wind speed at hub height decreased by up to 3 $m\,s^{-1}$ (figure 5(b)) and the temperature was slightly higher (the slope is nearly 1 but the intercept is 0.16 °C in figure 5(a) due to the wind farms. As such, the negligible increase in O_3 concentrations at the Mid-Atlantic wind farms may be due to the combination of this slight increase in temperature in the lower rotor area of the wind turbines and the reduced wind speed near the ocean surface. We will verify this hypothesis in the next section 4.2. In addition, given that VOCs are key precursors to O_3 formation, we examined changes in VOC concentrations to assess their role in the slight O_3 changes. We found a minimal rise in VOC levels (up to 0.02 ppb, not shown) potentially caused by a reduction in the wind speed. This increase likely plays a role in the slight O_3 increase within the farms, suggesting its contribution to the overall trend.

The next pollutant of interest is $PM_{2.5}$. When LWC is not included, which is the approach that is closest to measurements, as discussed in section 3, no significant differences (i.e. less than 0.1 $\mu g\,m^{-3}$) are detected between the $PM_{2.5}$ concentrations in the WithFarms and the Control cases (not shown). This finding is in agreement with Ruan *et al* [41] where a 0.067 $\mu g\,m^{-3}$ (0.08%) increase in PM2.5 was found. On the other hand, when LWC is included, the three-month average concentration difference is nearly zero over the entire domain, except for an area between the two largest WEAs offshore of New Jersey exhibiting a slight increase of less than 0.85 $\mu g\,m^{-3}$ (figure 3(b)), corresponding to less than 3% of the NAAQS. Because the average concentrations of $PM_{2.5}$ measured along the coast are well below the NAAQS (figure 2(b)) and because the only slight increase in the simulations was found to occur offshore and only when LWC was included, we do not expect that the WEAs will impact $PM_{2.5}$ significantly. This is also confirmed by the scatter plot analysis (figure 4(b)), where the slope of the linear regression between the $PM_{2.5}$ concentrations with and without the wind farms is one, which suggests no significant enhancement.

We find that NO_2 and SO_2 are not significantly impacted by the presence of the wind farms in terms of either the three-month averages (not shown in figure 3 because the changes are not detectable) or the scatter plots (figures 4(c) and (d)).

In summary, due to the dominant southwesterly wind direction in the region in the summertime [20], changes in meteorology and air pollutant concentrations predominantly remain in the offshore regions and do not reach the coastline, on average.

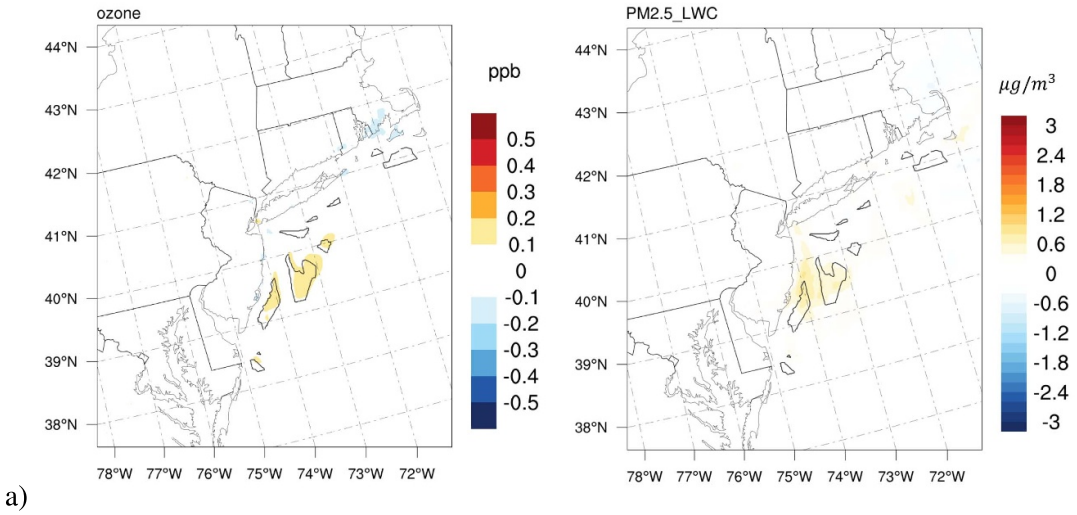

Figure 3. Changes induced by the offshore wind farms (WithFarms minus Control simulations) in the three-month average concentrations of: (a) 8 h average O_3 (ppb), and (b) 24 h average $PM_{2.5}$ with LWC ($\mu g\,m^{-3}$). The average concentration changes of $PM_{2.5}$ without LWC, NO_2 and SO_2 are below 0.1% of their respective NAAQS and therefore are not shown here.

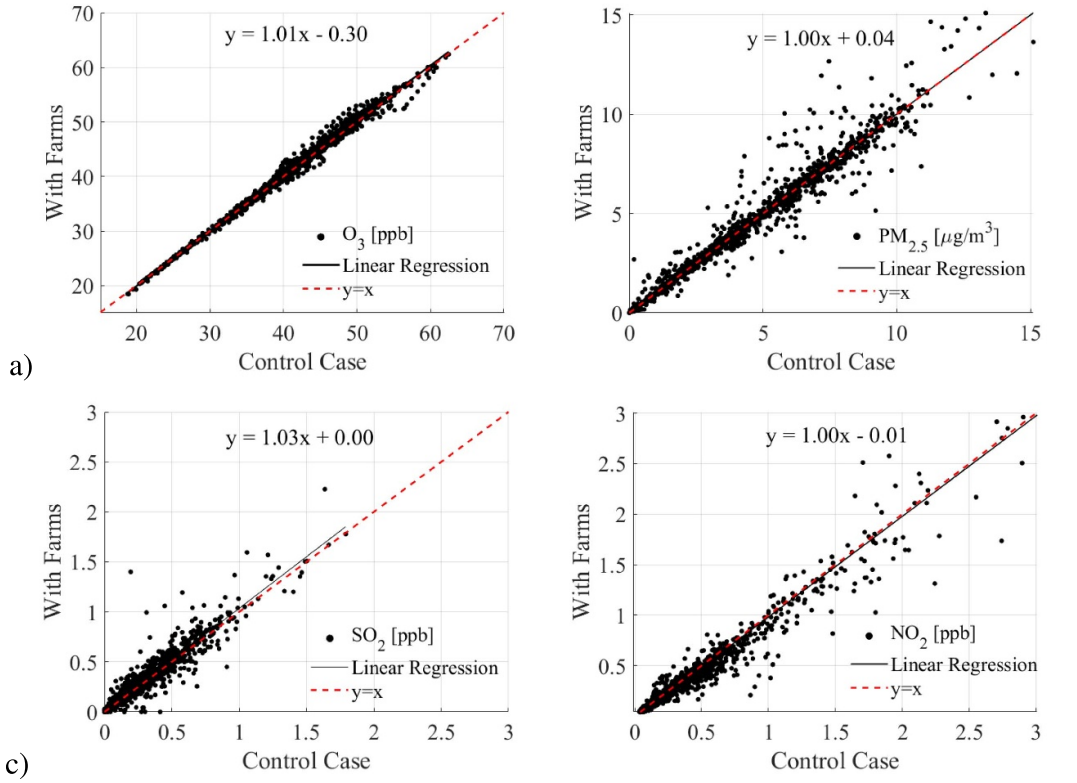

Figure 4. Scatter plots of results from the WithFarms case (y-axis) and the Control case (x-axis) for: (a) O_3 (ppb), (b) $PM_{2.5}$ ($\mu g\,m^{-3}$), (c) SO_2 (ppb), and (d) NO_2 (ppb), at the location near the NJ wind farm shown by the star in figure 1.

4.2. Ozone case study: 20 June 2018

Although the three-month average ozone concentrations are nearly unaffected by the BOEM-planned wind farms, occasional high-ozone episodes may occur due to specific meteorological and/or photochemical conditions. It is important to assess whether significant changes in ozone concentrations may occur at all during the summer and what conditions may be associated with them. We choose to focus on O_3 only because it is the only criteria pollutant that is still in non-attainment in several regions along the coast and therefore even a few ppb of difference due to the WEAs may cause a violation of the NAAQS during a high-ozone event.

Here we assess the impact of wind farms on 20 June, the day that was most significantly influenced by

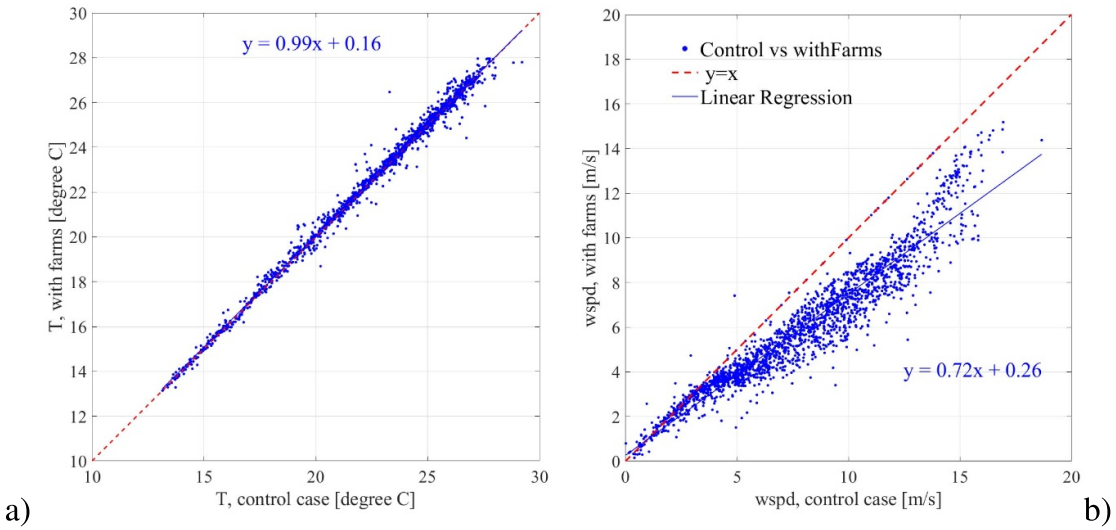

Figure 5. Scatter plots of results from the WithFarms case (y-axis) and the Control case (x-axis) for: (a) temperature (°C) and (b) hub-height wind speed (wspd) (m s^{-1}), at the location near the NJ wind farm shown by the star in figure 1.

the presence of wind farms, with observed increases of up to approximately 2 ppb in O_3 concentrations downwind of the NJ WEAs (figure 6(a)). We also detect a very small decrease in O_3 along the coast and inland away from the wind farms (up to 0.6 ppb). However, it is essential to exercise caution in interpreting the very minor decrease away from the farms, as figure 6(a) represents the average over an individual day rather than the entire three-month duration. This distinction introduces the possibility of model noise influencing the WRF results, potentially leading to non-physical alterations, as discussed in previous studies [20, 40]. Consequently, while weak wind accelerations at the edges of wind farm wakes are well known to occur (e.g. figures S3 and S4 of [20]) and could lower local pollutant concentrations, the minor changes in O_3 concentrations far away from the wind farms may or may not be attributed to numerical noise. We note that numerical noise is introduced by the WFP in WRF, not by CAMx.

Looking at the wind speed and temperature vertical profiles at the location marked with a star in figure 1 on 20 June for the WithFarm and Control cases (figures 6(b) and (c)), the ambient temperature is higher in the lower part of the rotor area by up to 0.35 °C (on average) in the presence of the wind turbines. Similarly, the wind speed is constantly lower at all levels due to the wind turbines' wakes. This is consistent with the previously mentioned hypothesis that the slight ozone increase is associated with this small increase in temperature and reduction in wind speed right between the rotor and the ocean surface.

5. Conclusions

The goal of this study is to investigate if the changes in meteorology caused by offshore wind farms could alter the air quality along the coast. The impacts of the large-scale U.S. offshore wind farms on the meteorology of the coastal areas were discussed in our previous article [20]. Here, we present the direct effects of those meteorological changes on the air pollutant concentrations in the region. We used the WRF model, equipped with the Fitch WFP, to simulate the meteorology along the East Coast during three months in the summer of 2018 under two separate scenarios, i.e. including and excluding the large-scale BOEM-planned wind farms. Then we used the WRF model outputs as meteorological inputs for the CAMx air quality model to simulate the impacts on ground-level ozone, particulate matter, sulfur dioxide, and nitrogen dioxide, all four of which are criteria pollutants regulated by the EPA. We used the CAMx air quality model with the same horizontal and vertical grid resolution as the parent domain in the WRF model; i.e. 315 × 300 grids with a 4 km grid resolution.

Our findings suggest that, overall, the impacts of offshore wind farms on air pollution levels are expected to be negligible. On average, O_3 increased at most 0.2 ppb within the areas of the offshore wind farms over the ocean. This amount is less than a 1% increase in O_3 concentration. At worst, during the most impacted day over the three-month simulations (20 June), O_3 increased by up to 2.5 ppb in the wake

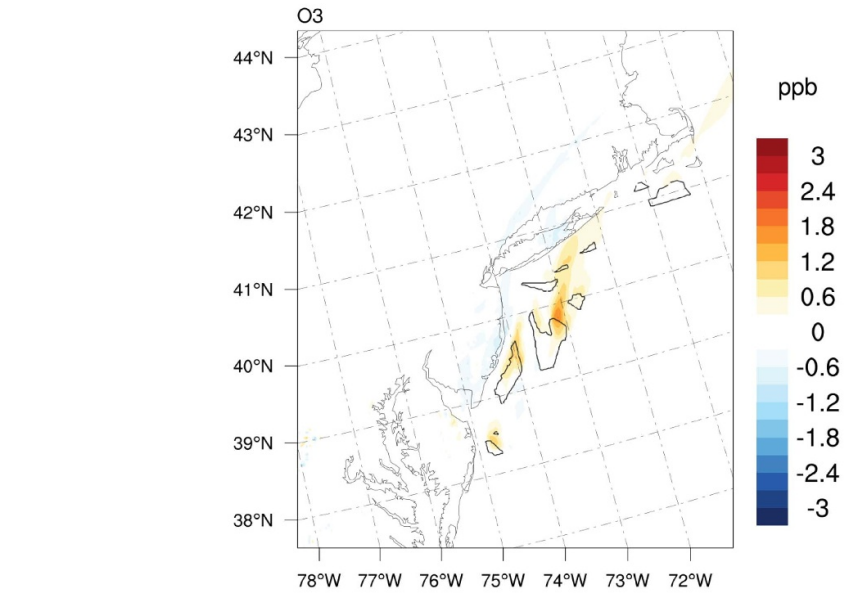

Figure 6. (a) Diurnal average of the changes in O_3 concentrations due to the offshore wind farms (WithFarms—Control) on 20 June 2018, with the highest impact on O_3 concentrations within the three-month simulation period. The diurnal average of the vertical profiles is shown for (b) temperature, and (c) wind velocity on 20 June 2018. The vertical profiles are calculated at the location shown by 'star' in figure 1.

of the NJ wind farm over the Atlantic. The increase in O_3 may be due to the slight warming in the rotor area of the wind farms and the lower wind speed. In addition, $PM_{2.5}$, SO_2, and NO_2 concentrations remained basically unchanged in the entire domain. Overall, we conclude that the impacts of the meteorological changes due to the large-scale offshore wind farms on the East Coast of the U.S. will be imperceptible to humans.

Data availability statement

The data for the model setup is available in the GitHub repository at: https://github.com/golbazimaryam/ShippingEmissionsAndAirQuality.

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Author contributions

M Golbazi contributed to the study design, writing the manuscript, setting up and carrying out simulations, and data analysis. C L Archer contributed to

the project design, editing the manuscript, and data analysis.

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Conflict of interest

The authors declare that they have no conflict of interest.

Ethical statement

As this study did not include any human subjects, ethical approvals were not required.

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