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Impacts of Aviation Emissions on Near-Airport Residential Air Quality

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ABSTRACT: Impacts of aviation emissions on air quality in and around residences near airports remain underexamined. We measured gases (CO, CO_2, NO, and NO_2) and particles (black carbon, particle-bound aromatic hydrocarbons, fine particulate matter ($PM_{2.5}$), and ultrafine particles (reported using particle number concentrations (PNC) as a proxy)) continuously for 1 month at a residence near the Logan International Airport, Boston. The residence was located under a flight trajectory of the most utilized runway configuration. We found that when the residence was downwind of the airport, the concentrations of all gaseous and particulate pollutants (except $PM_{2.5}$) were 1.1- to 4.8-fold higher than when the residence was not downwind of the airport. Controlling for runway usage and meteorology, the impacts were highest during overhead landing operations: average PNC was 7.5-fold higher from overhead landings versus takeoffs on the closest runway. Infiltration of aviation-origin emissions resulted in indoor PNC that were comparable to ambient concentrations measured locally on roadways and near highways. In addition, ambient NO_2 concentrations at the residence exceeded those measured at regulatory monitoring sites in the area including near-road monitors. Our results highlight the need for further characterization of outdoor and indoor impacts of aviation emissions at the neighborhood scale to more accurately estimate residential exposures.

■ INTRODUCTION

In 2018, 10 million flights carrying one billion passengers flew into or out of airports in the United States (US).[1] Over the next 25 years, flight operations and enplanements in the US are projected to grow annually at the rate of 1 and 2%,[2] respectively, and a similar outlook is expected worldwide.[3] To meet this growing flight demand, in the last two decades over half of the 35 busiest airports in the US have undergone airfield expansions to increase their capacity.[4] These trends are of significance to the health of millions of people who live or work near airports and are thereby regularly exposed to noise and air pollution originating from aviation activity. For example, increased rates of adverse health outcomes ranging from hypertension,[5−13] cardiovascular disorders,[6,14−16] birth outcomes,[17,18] respiratory diseases,[19] and learning deficit in children[20−22] have been observed near airports. Furthermore, implementation of the Next Generation Air Transportation System,[23] which guides airplanes on precise paths via satellites, has narrowed the flight paths and lowered landing altitudes, concentrating the impacts further in certain communities.

Recently, the impacts of aviation emissions on ground-level ambient ultrafine particle (UFP; aerodynamic diameter < 100 nm) concentrations were found to extend over unexpectedly large areas near airports and in particular along flight paths.[24] For example, elevated particle number concentrations (PNC) were reported downwind of major international airports as far as 7 km near Amsterdam, 7.3 km in Boston, 18 km in Los Angeles, and 22 km in London.[25−29] UFPs are emitted at high rates by jet aircraft[30] and linked to increased rates of hypertension and cardiovascular morbidities.[31,32] However, UFPs do not contribute significantly to mass in the fine particle range and are not routinely monitored, in part due to a lack of ambient air quality standards. Therefore, they present the possibility of being an additional important confounder for near-airport epidemiological investigations.[33,34] For example, Wing et al.[35] found that UFP exposure was. independently associated with adverse birth outcomes in the vicinity of Los Angeles International Airport. Similarly, black carbon (BC) and oxides of nitrogen, which are also emitted at high rates by aircraft[30,36−38] and have recognized adverse cardiovascular effects,[39] are also elevated near airports.[24,25] Some near-airport epidemiological studies have accounted for confounding pollutants, like fine ($PM_{2.5}$[16]) and coarse particulate matter (PM_{10}[15,40,41]), ozone,[16] and NO_2,[41] but by using regional-scale central monitor data or predictive models that only account for larger-scale spatial patterns and ground-transportation emissions. Confounding co-exposure to aviation-origin emissions

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themselves remains unaccounted for, limiting the causal interpretation of the epidemiological results.

Moreover, research on near-airport air quality has been limited to ambient (outdoor) observations to date.[24] The extent and conditions under which aviation emissions infiltrate residences and impact indoor air quality remain largely unaddressed. We found only one study that reported on residential infiltration of aviation emissions.[42] In that study, 16 homes in Boston (MA), which were selected primarily for assessment of highway impacts on indoor air quality, were found to contain higher PNC indoors when the residences were downwind of the Logan International Airport. This study did not quantify infiltration rates due to the lack of concurrent outdoor and indoor measurements. Further, no studies have investigated the influence of meteorology and aviation activity on infiltration or quantified impacts of aviation emissions on indoor air quality.

In this study, we concurrently monitored outdoor and indoor air pollutant concentrations in a near-airport residence to assess the influence of temporal factors including time of day, meteorology and aviation activity intensity, and operation type (landings and takeoffs). We studied a residence of a common architectural style and vintage in Winthrop, MA, a community that is significantly impacted by the Logan International Airport. About a third of the Winthrop's population of 17 500 lives within the 60 dB noise impact zone (an annual average of cumulative 24 h day and night noise exposures with a 10 dB night-time penalty).Ours is the first study to detail the disproportionate impact of overhead landing jets on residential outdoor and indoor air quality.

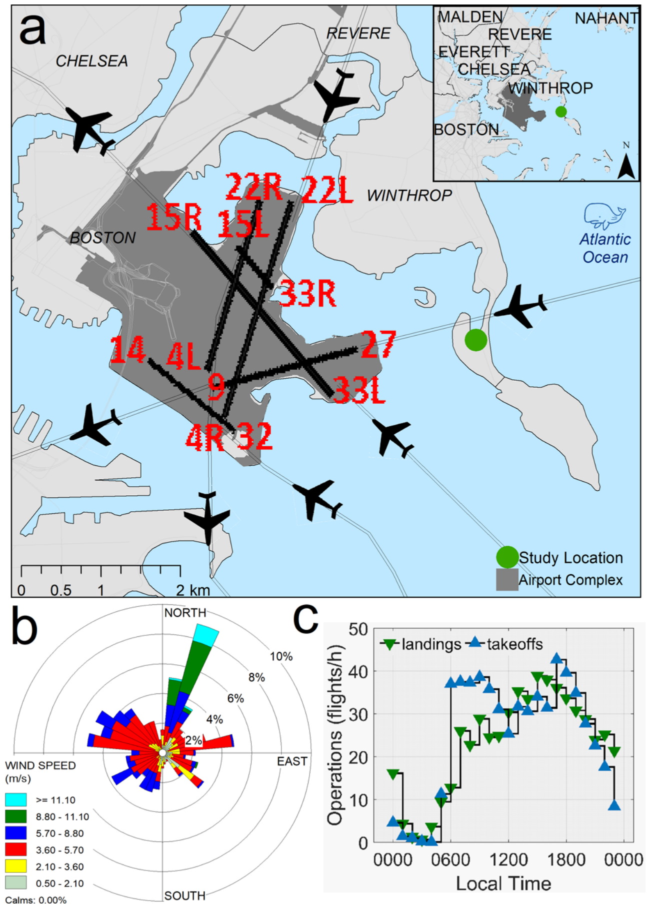

Figure 1. (a) Monitoring the site location and flight trajectories for preferred runway configurations for jets during SW and NW winds at the Logan Airport. (b) Windrose and (c) diurnal flight activity for the study period.

■ METHODS

Airport Description. The General Edward Lawrence Logan International Airport is located 1.6 km east of downtown Boston (Figure 1). It has six runways and supports ~1000 operations per day (combined landings and takeoffs [LTO]). For each wind-direction quadrant, the airport has a '*preferred runway configuration*' consisting of a subset of runways (three out of the six runways), as shown in the Supporting Information (SI), Figure S1, to which operations are preferentially directed. In the US, the naming convention on runways is such that they represent the numerical heading in tens of degrees of the planes using the runways. For example, planes taking off or landing on runway 27 at the Logan airport head ~270° true north, while planes taking off or landing on runway 4 head ~40° true north.

Residential Air Quality Monitoring and Instruments. Monitoring was conducted from August 23 to September 23, 2017, at a residence in Shirley Point, Winthrop located 1.3 km from the eastern end of runway 9/27 (Figure 1). Jets descend overhead of the residence at an elevation of ~75−100 m. The residence is located in a suburban neighborhood with only one major collector/arterial road within a 1 km radius, and the road leads to a dead end and thus has very limited vehicular traffic (Figure S2a). Outdoor monitoring was performed using the Tufts Air Pollution Monitoring Laboratory (described in detail elsewhere[43]), which was parked in the driveway on the northern side of the property. Outdoors, a suite of gaseous and particulate pollutants were measured including particle number concentrations (PNC, measured at 1 s resolution using a TSI (MN) Condensation Particle Counter 3783 [CPC, d_{50} = 7 nm]), black carbon (BC), fine particulate matter ($PM_{2.5}$), particle-bound polycyclic aromatic hydrocarbons (PBPAH) for PAH-containing particles ≤1.0 μm), carbon monoxide (CO), and oxides of nitrogen (NO, NO_2, and NO_X) (see Table S1 for details of the instruments used). To limit disturbance to the residents (i.e., due to noise from the monitoring equipment), indoor measurements were restricted to PNC using the same make and model CPC as outside. It was placed in the first-floor living room. Weekly maintenance of the instruments included flow checks, clock resets, and data download.

Residence Characteristics and Ventilation Practices. The residence, built in 1920, is a two-story, two-bedroom, 1700 ft^2 colonial wood-frame house that is typical of the architectural style of the neighborhood. It does not have a centralized ventilation system (neither AC nor fans) and neither the kitchen nor the single bathroom is equipped with exhaust fans. It has eight double-hung windows, four picture windows, two inoperable windows, a front door, a back door, and a sliding glass door. In the early 1990s, all of the windows, the front door, and the sliding glass door were replaced with new, tighter versions as part of Massport's Residential Sound Insulation Program.[44] This is a voluntary program where owners of residences located within the 65 dB DNL threshold area can apply for noise reduction measures. Therefore, this residence may have lower air exchange rates under closed window conditions than residences without soundproofing. New storm windows and storm doors were also added at this time.

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Environ. Sci. Technol. XXXX, XXX, XXX−XXX

Prior to the start of monitoring, we deliberately did not discuss ventilation practices (or instruct residents to modify or not modify current practices) so as not to influence their behavior during the month of monitoring. Following the monitoring period, the residents were surveyed *post hoc*[45] on cooking practices, fan and air conditioner use, and window openings during the month-long study. On the survey, the residents indicated that on weekdays windows were opened minimally during the day (~2 h) and in the evening (~1 h), while on weekends, 3−5 windows were typically opened for 2−5 h during the day and 1−2 h in the evening (1800−2300 h). At night (after 2300 h) on both weekdays and weekends, all windows were closed and as many as three window-mounted AC units were operated to provide cooling. Also, the residents (two full-time working adults) indicated that they cooked infrequently.

Regulatory Monitoring Sites and Other Sites. To provide perspective on near-airport observations, we also compare concentrations of pollutants measured near the airport with those measured at regulatory monitoring sites and in near-highway neighborhoods in the Boston area. Data from five proximal regulatory monitoring sites operated by the Massachusetts Department of Environmental Protection[46] in Suffolk and Essex counties were obtained. For ease of interpretation, we refer to these sites by their distinguishing features. The sites are as follows: (a) a site on the shoulder of Interstate I-93N and 6 km SW of the airport (referred to as *adjacent-highway*); (b) a near-roadway site at the intersection of five streets and 100 m N from Interstate I-90 and ~6 km W of the airport (*near-roadway*); (c) a site in downtown Boston 3.5 km W of the airport (*downtown*); (d) a site located 7.5 km WSW of the airport that is considered indicative of the neighborhood scale (*urban-background*); and (e) a site 13 km NNE of the airport in Lynn, MA, that is considered indicative of regional-scale air quality (*regional-background*). Traffic volumes (annual average daily traffic estimated from the regional planning commission[47]) in the 1 km area around these sites are shown in Figure S2b−f in Table S2.

The air quality monitoring instruments used at regulatory sites are listed in Table S2. We used federal equivalent method instruments to measure CO and oxides of nitrogen at the near-airport site. For $PM_{2.5}$, we used an optical sensor instead of federal reference/equivalent methods; this nephelometer tends to read higher than federal reference/equivalent methods, is sensitive to relative humidity, which we do not correct for, and requires gravimetric calibration to local aerosol for data to be comparable to regulatory data.[48] Thus, we do not discuss absolute $PM_{2.5}$ concentration differences between the near-airport residence and the regulatory sites and limit interpretation to broad trends.

Because ultrafine particles are not a regulated air pollutant in the US, PNC is not routinely monitored at the regulatory sites by state or federal agencies. Comparable PNC data were available from the Tufts UFP Monitoring Network (TUMN), which uses the same CPCs as we used at the near-airport residence (TSI model 3783). Data were available from two locations: first, the roof of a three-story building in Chelsea, 4.0 km northwest from the airport, for the entire study duration (August 23−September 23, 2017) and second, from a station collocated at the urban-background regulatory site for August 23−September 09, 2017.

Data Acquisition and Statistical Analysis. Meteorological data collected at the airport (KBOS) were obtained from the National Centers for Environmental Information[49] and aggregated to hourly resolution. Regulatory monitoring site data was obtained from EPA's AirData websites https://www.epa.gov/airdata and https://aqs.epa.gov/api at hourly average resolution. Measured pollutant data were aggregated to hourly resolution and aligned with the meteorological and regulatory data.

Data on flight activity at the airport were web-scraped from https://secure.symphonycdm.com, a public portal for tracking flight activity at the airport. A coordinate grid was established for each runway, and when a plane entered or exited, a grid, it was counted as having landed or taken off, respectively. Data was extracted at 30 s intervals and aggregated to the hour. To check for errors in the automated methodology, flight activity was also replayed and tracked manually for 5 h (three busy hours with >2 operations/minute and two more hours with <0.5 operations/minute) (Table S3); scraping/automated extraction underestimated operations by 0−3% in busy hours and 0% in other hours. Detailed flight activity logs including idling and taxiing times for airplanes on the tarmac were unavailable to us.

Statistical analysis was conducted in MATLAB 2018. Nonparametric statistics were used because the pollutant data were non-normally distributed; differences were tested using the Wilcoxon rank-sum test (significance threshold $p < 0.05$), and the Spearman's rank correlation coefficients (r_s) are reported. Extreme outliers were defined using Tukey's fences,[50] i.e., three times the interquartile range, and excluded from indoor-to-outdoor (I/O) ratio analysis (amounting to 0.007% of data during impact sector and 4.9% of the data during other winds). As a check, all extreme outliers were found to exceed unity, indicating that indoor concentrations exceeded outdoor concentrations likely due to indoor sources.

■ RESULTS AND DISCUSSION

Flight Activity Patterns. SW-NNW winds orient the residence downwind of the airport. During these winds, landings occurred mostly over the water and takeoffs occurred mostly over the land (Figure 1a shows flight trajectories). For example, when winds are from the S-W (180−270°), the predominant wind direction (WD) in the Boston area during summer, jets are preferentially directed to land on 22L (heading 214.6°) and 27 (heading 271.5°) and takeoffs are directed to occur on 22L and 22R (heading 214.6°). When winds are from the W-N (270−360°), flights are preferentially directed to land on runways 27, 32 (heading 320.6°) and 33L (heading 330.1°) and takeoff from 27 and 33L. During the study, 100% landings and 100% takeoffs occurred on preferred runways for 62 and 48% of the hours, respectively, and >50% of the operations occurred on preferred runways 70% of the hours. Takeoffs were far more frequently directed to nonpreferred runways than landings (e.g., during SW and NW winds, ~15% of takeoffs occurred on nonpreferred runways compared to <5% of landings). The windrose and flight activity for the study duration are shown in Figure 1b,c. Overall, we observed 1.2 times as many flights during evening peak rush hour (1700−1800) than during morning peak rush hour (0900−1000). The hours of 0100−0600 were the least busy due to night-time flight restrictions (Figure 1c).

Wind Direction and Pollutant Patterns. The WSW-N sector (247.5−360°) stands out in the bivariate polar plots as the sector associated with the highest PNC (Figure 2a,b), a trend also reflected by most of the other pollutants (Figure

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Environ. Sci. Technol. XXXX, XXX, XXX−XXX

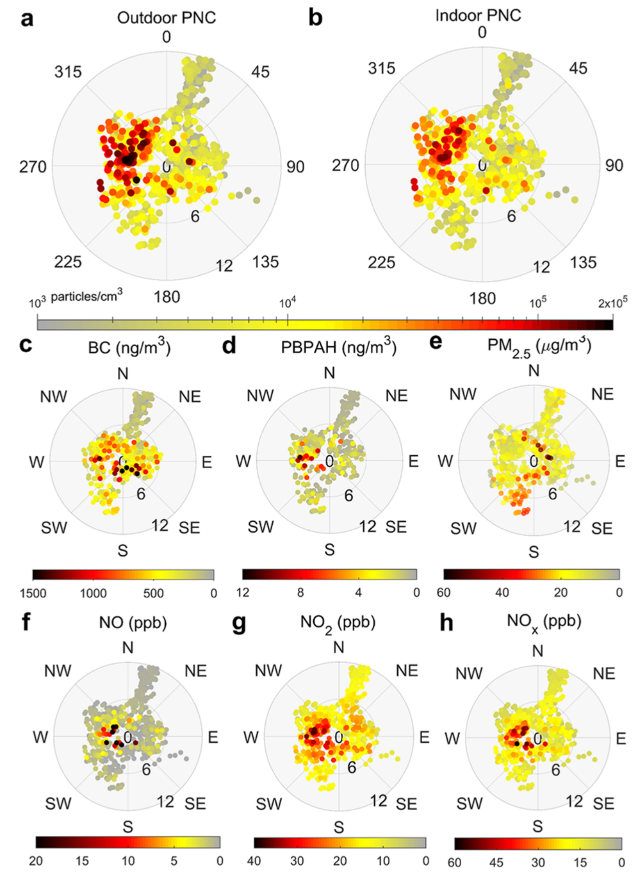

Figure 2. Polar plots of hourly average pollutant concentrations versus wind direction and speed. (a) Outdoor and (b) indoor particle number concentrations (particles/cm^3) and (c−h) hourly average outdoor concentrations of other pollutants. Radial axis labels placed along 135°show wind speed in m/s.

2c−h). During these winds, the residence was downwind of the airport complex as well as flight trajectories for the runways preferred during westerly winds. We identified this sector as the impact sector (247−360°), similar to other works.[27,42,51,52]

Non-impact-sector winds (i.e., winds from 0 to 246.9°) oriented the monitoring site upwind of the airport and were further subdivided into winds from over-the-ocean (0−112.5°) and over-the-land (112.6−246.9°).

Significantly higher concentrations of PNC, oxides of nitrogen (NO, NO_2, and NO_X), CO, BC, and PBPAH were observed during impact-sector winds compared to non-impact-sector winds (Table 1). Fold elevation, or the ratio of mean concentration for all hours of impact-sector winds to the mean concentration for all hours of non-impact-sector winds, was highest for PNC (Table 1): PNC were 4.8-fold elevated outside and 4.2-fold elevated inside the residence. Fold elevation was lower for other pollutants. BC was 1.3-fold elevated and PBPAH was 1.8-fold elevated. NO, NO_2, and NO_X were 1.9, 1.2, and 1.2-fold elevated, respectively (n.b., the difference between means was much greater for NO_2 (2.7 ppb) than for NO (0.8 ppb)). Fold elevation was lowest for CO, 1.1-fold. Only $PM_{2.5}$ concentrations were not elevated during impact-sector winds relative to non-impact-sector winds. Higher $PM_{2.5}$ concentrations were observed when winds were from the S−SW, a pattern consistent with that observed at vicinal regulatory monitoring sites (Figure S3a) and associated with long-range transport of aerosols from regional sources upwind.

Generally, when winds were from over-the-ocean, pollutant concentrations were lower; the lowest levels occurred during a 3.5-day-long storm event (mid-day 19 September−23 September, 2017), during which winds were high and from the NNE (see Figures 1c and 2). Table S4 summarizes concentrations for non-impact-sector winds further split into over-the-ocean and over-the-land winds.

Diurnal Patterns. PNC diurnal patterns during impact-sector winds were very distinct from those for other pollutants and distinct from PNC diurnal patterns during non-impact-sector winds. As shown in Figure 3, PNC increased steadily from 1600 to 2300 h to levels that were far higher than those at any other time of the day and decreased precipitously with a drop in flight activity, in particular, after 0100 h. The late-evening (2000−2300 h) average exceeded the morning (0600−1100 h) average by a factor of three (80 000 ±

Table 1. Statistics for Hourly Averaged Pollutant Concentrations during Monitoring (23 August−23 September, 2017)

	n (hours of data)		mean (± st. dev.)			median (IQR)		Wilcoxon rank-sum test statistics[a]	
pollutant	impact sector	non-impact sector	impact sector	non-impact sector	fold elevation	impact sector	non-impact sector	*p*-value	*z*-value
PNC indoors (number/cm^3)	261	469	25000 ± 27000	6000 ± 8000	4.2	13 000 (6000−32 000)	4000 (2000−7000)	<0.05	14.1
PNC outdoors (number/cm^3)	255	484	38000 ± 42000	8000 ± 15000	4.8	17 000 (7000−55 000)	4000 (3000−7000)	<0.05	15.0
PNC I/O ratio	255	469	0.77 ± 0.27	0.83 ± 0.23		0.78 (0.60−0.91)	0.81 (0.69−0.95)	<0.05	−3.7[b]
BC (ng/m^3)	141	370	390 ± 230	300 ± 250	1.3	330 (230−530)	250 (130−390)	<0.05	5.0
PBPAH (ng/m^3)	159	229	1.8 ± 1.9	1 ± 1.1	1.8	1.1 (0.7−2)	0.6 (0.4−1.2)	<0.05	6.9
$PM_{2.5}$ (μg/m^3)[c]	251	419	11 ± 4	15 ± 7		11 (8−13)	13 (11−17)	1	−7.1
NO (ppb)	252	419	2 ± 3	1 ± 2	1.9	1 (0−2)	0 (0−1)	<0.05	7.3
NO_2 (ppb)	252	419	17 ± 7	14 ± 5	1.2	15 (12−21)	13 (11−16)	<0.05	5.2
NO_X (ppb)	252	419	18 ± 8	15 ± 6	1.2	16 (13−22)	14 (11−17)	<0.05	5.6
CO (ppb)	196	401	220 ± 50	200 ± 60	1.1	210 (180−240)	180 (150−230)	<0.05	5.3

[a]One-sided hypothesis test, where the alternative hypothesis states that the median of the impact sector is greater than the median of other winds.
[b]One-sided hypothesis test, where the alternative hypothesis states that the median of other winds is greater than the median of the impact sector.
[c]Factory calibration based.

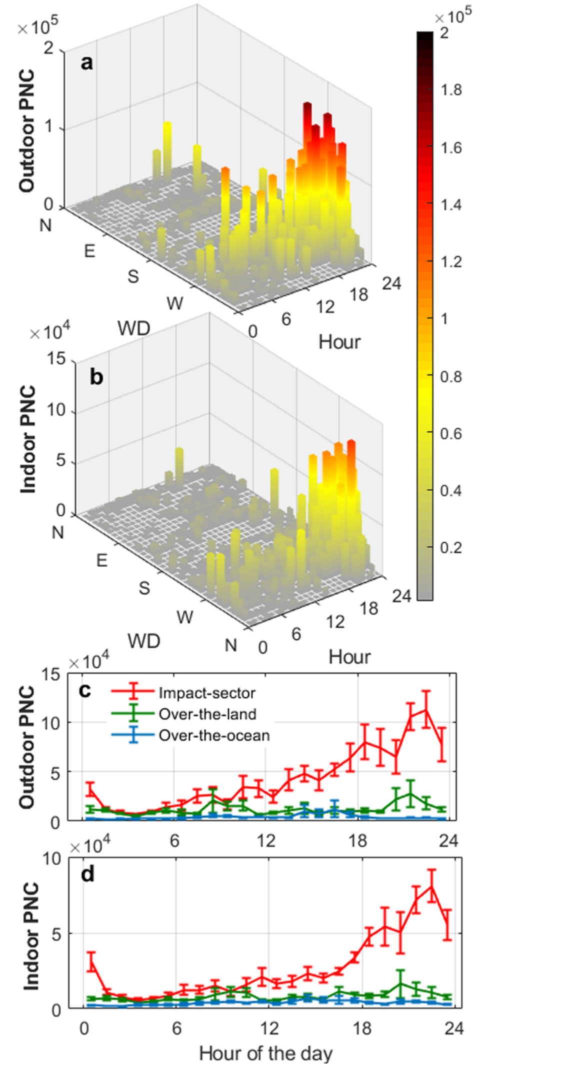

Figure 3. Outdoor (a) and indoor (b) PNC patterns with respect to wind direction (WD) and hour of the day; data were binned into 36 10°-wide WD and 24 hourly bins, resulting in an unequal distribution of data per bin but reflecting the natural frequency of WD during the monitoring period. Diurnal trends of outdoor (c) and indoor (d) PNC for impact-sector and other winds. Error bars show the standard error. Note the difference in the y-axis scale for outdoor versus indoor PNC.

51 000 versus 25 000 ± 26 000 particles/cm^3) even though total flight operations were only 15% higher in the evening relative to the morning. This indicates that the late-evening PNC increase was promoted by factors other than a proportional increase in flight activity. We also observed a pronounced late-evening PBPAH peak during impact-sector winds. PBPAH are emitted directly in aircraft exhaust, but they can also form due to condensation of semivolatile PAH on particles in the atmosphere.[37] The highest ratio of PBPAH concentration to BC concentration (BC is also emitted directly in aircraft exhaust and is a relatively inert pollutant) also occurred in the late evening hours during impact-sector winds (Figure S5). Lack of detailed tarmac-level activity data (idling and taxiing times) and chemical composition precludes an explanation for the late-evening PNC increase we observed. For example, the increase could have derived from greater airplane idling and other low-thrust operations during evening hours; low-thrust operations like idling have a higher PNC emission index (number of particles/kg fuel burnt) than high-thrust operations.[53] Greater knowledge of how plumes chemically evolve as they are transported from airplanes to downwind receptor areas near airports could help to better explain our findings.

Other than PNC, all of the pollutants had bimodal diurnal concentration profiles during impact-sector winds and the magnitude of morning and evening peaks were comparable except for NO, where the morning peak concentration was about 3-fold higher than the evening peak concentration, and NO_2, where the average concentration in late-evening exceeded the morning average by 1.3-fold (Figure 4).

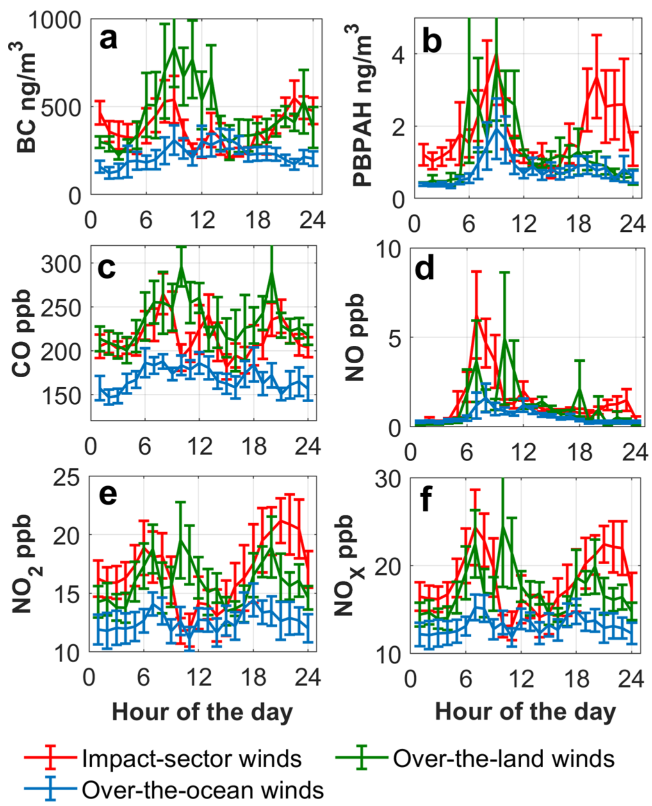

Figure 4. Diurnal trends of hourly averages of outdoor pollutant concentrations for the monitoring period during impact-sector and other winds. Error bars show the standard error. See Figure S3b for $PM_{2.5}$ and Figure S6 for CO_2.

In comparing non-impact-sector/over-the-land winds, non-impact-sector/over-the-ocean winds, and impact-sector winds, several key observations emerge. First, during over-the-ocean winds, the concentrations of pollutants were consistently and expectedly the lowest compared to other wind sectors. Also, upon examination of over-the-ocean diurnal patterns, there were small coincident peaks of PBPAH, BC, NO, and NO_2 in the morning. The few upwind air pollution sources in this sector include marine vessels, activities at Deer Island where the Deer Island Wastewater Treatment Plant is located, and traffic on roadways near the monitoring site; it is possible that these sources were responsible. Second, during over-the-land winds, the concentrations of pollutants were lower than impact-sector winds, except BC and CO. During morning to mid-day hours (0500−1300), BC concentrations during over-the-land winds were substantially higher than during impact-

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Environ. Sci. Technol. XXXX, XXX, XXX−XXX

sector winds (650 ± 120 versus 390 ± 110 ng/m^3). The diurnal profiles for CO during over-the-land and impact-sector winds were similar and concentrations were only moderately different (230 ± 60 versus 220 ± 50 ppb); the evening peak coincided with the ground-traffic rush-hour period (1700–2000 h), indicating the influence of primary vehicular emissions from the Boston area at this monitoring site. Third, no distinct diurnal pattern was observed for $PM_{2.5}$ (Figures S3b and S7). Fourth, the diurnal pattern for CO_2 was similar for all three wind sectors (Figure S6). Finally, during both impact-sector and non-impact-sector winds, the lowest concentrations of pollutants were observed during 0200–0500 h when flight activity was minimal (Figure 1c) and during the warm afternoon hours when convective mixing was greatest. Correlations between pollutants at hourly time resolution are discussed in the SI (Figure S7).

Particle Infiltration. Indoor diurnal PNC patterns were nearly identical to outdoor PNC patterns (Figure 3), indicating that there was substantial infiltration of outdoor particles into the residence. Time-series plots based on 1 s measurements indicate that infiltration occurred rapidly, on the order of minutes (Figure S8). Overall, indoor PNC during both impact-sector and non-impact-sector winds were only ∼25% lower than outdoor PNC but there were modest wind-sector differences in the I/O ratios. I/O ratios were significantly ($p < 0.001$) lower during impact-sector winds compared to other winds: 0.77 ± 0.27 during impact-sector winds compared to 0.82 ± 0.23 during other winds (Figure S9a). In addition, the I/O ratios were generally negatively correlated with outdoor concentrations (Figure S9e), but more strongly so for impact-sector winds ($r_s = -0.49$, $p < 0.0001$) than non-impact-sector winds ($r_s = -0.32$, $p < 0.001$). These results are consistent with the expectation that the I/O ratios should be lower for particle mixtures dominated by smaller particles (like aircraft emissions[26,54]) because they have lower penetration rates or higher diffusional losses through cracks.[55] But the differences are modest, and coincidental influence of unquantified factors, like irregular window opening, cannot be ruled out.

Flight Activity on Preferred Runways and Pollutant Patterns during Impact-Sector Winds. Pollutant concentrations and correlations with flight activity strongly depended on the operational runway configuration. The highest correlations between ambient pollutant concentrations and total flight activity (combined landings and takeoffs per hour; LTO/h) occurred when the preferred runway configuration for impact-sector winds was used. For these conditions, all pollutants except $PM_{2.5}$ were positively correlated with total flight activity (r_s ranged from 0.31 to 0.57 for landings and 0.28–0.54 for takeoffs (Figure S12)). In contrast, flight activity on nonpreferred runways, even during impact-sector winds, was negatively correlated with pollutant concentrations although the monitored residence was still downwind of the airport (r_s ranged from −0.48 to −0.17 for landings and −0.45 to −0.22 for takeoffs). Correlation coefficients for all pollutants are shown in Figure S12.

Further, whether jets were landing or taking off at a particular runway made a remarkable difference on the downwind impacts. This point is illustrated in Figure 5a, which shows outdoor and indoor PNC (1 s resolution data) and the fraction of hourly flight activity on runways 27 and 33L. These are the two closest runways to our monitoring site. They are also preferred for operations during impact-sector winds (Figure 1a) and the majority of flight operations were

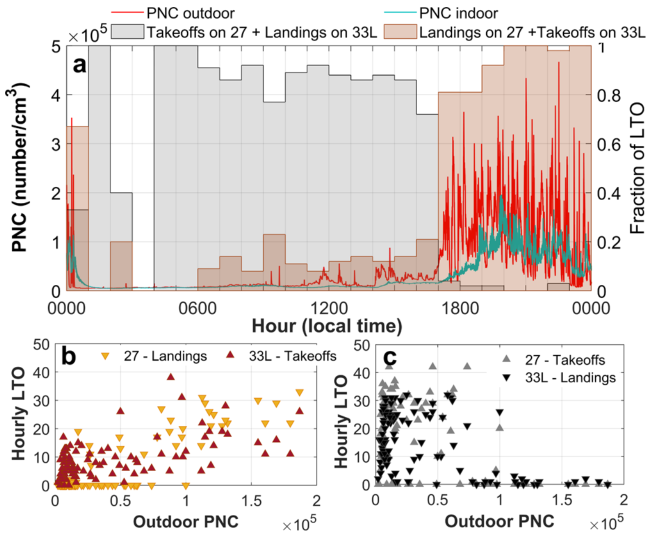

Figure 5. (a) 24-h time series of PNC and fraction of flight activity on runways 27 and 33L for a day of sustained impact-sector winds. (b, c) Scatter plots of hourly operations exclusively on runways 27 and 33L and outdoor PNC during impact-sector winds over the entire study period (n = 103 h). Figures S13–S16

conducted on these two runways during the 24-h period shown in Figure 5a. One key difference over the course of this day was that between 0400 and 1700 h, 70–100% of operations/hour occurred such that landings were on 33L and takeoffs on 27, but between 1700 and 0000 h the runway configuration switched and 80–100% of operations/hour occurred such that landings were on 27 (i.e., overhead of our monitoring site) and takeoffs on 33L. Concurrent with this switch at 1700 h, we observed recurrent PNC spikes that exceeded 100 000 particles/cm^3 and an overall increase in both outdoor and indoor PNC. Average outdoor PNC were 7.5-fold higher (121 000 ± 74 000 versus 16 000 ± 10 000 particles/cm^3) during 1700–0000 h than during 0400–1700 h. Likewise, average indoor PNC were similarly 7.7-fold higher (73 000 ± 31 000 versus 9 500 ± 3 400 particles/cm^3) during 1700–0000 h than during 0400–1700 h. Time series for other pollutants from the same 24-h period Figure 5 are shown in Figures S13–S16. Our results are consistent with reported observations of ground-level PNC spikes from descending plumes of landing jets under the landing trajectory up to 2.75 km from the runway.[26] It is noteworthy that pollutants are known to be entrained in the descending vortices from the jet wingtips[56] and the altitude of descending overhead jets at this residence (75–100 m) is below the expected planetary boundary layer height in summer. Examination of the hours in which flight activity occurred exclusively on runways 27 and 33L, i.e., n = 103 h, 40% of the impact-sector data set yielded similar results; data is shown in Figure 5b,c and statistics are discussed in the SI.

Comparison with Regulatory and Other Data sets. We compared our measurements from the near-airport residence to measurements collected during the month-long study period at five regulatory monitoring and two UFP monitoring locations in the Boston area. Locations for all sites are shown in Figure S17a, diurnal trends are shown in Figure S17b–h, and concentrations are summarized in Table S8.

The most interesting intercomparison was observed for oxides of nitrogen. Jet-engine exhaust emissions are highly

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Environ. Sci. Technol. XXXX, XXX, XXX−XXX

enriched in NO_2[38] relative to NO, while exhaust emissions from ground-transportation gasoline engines are primarily in the form of NO. NO can be oxidized within minutes to NO_2 in the presence of high ozone concentrations.[57] NO_2 concentrations at the near-airport residence were higher than those recorded at all of the regulatory monitoring sites including the ones adjacent to highways and busy roadways. Study-duration ambient average NO_2 at the residence was 15 ± 6 ppb (17 ± 7 ppb during impact-sector winds). This is ~40% higher than at the adjacent-highway (11 ± 7 ppb) and near-roadway (12 ± 8 ppb) sites, which are purposefully monitored to account for the highest exposures as part of EPA's and MassDEP's near-roadway network.[58] It was also nearly 2-fold higher than at the urban-background site (8 ± 7 ppb) and 7.5-fold higher than at the regional-background site (2 ± 3 ppb). In contrast, NO concentrations at the near-airport residence were lower than those at all regulatory sites except the regional-background site. Expectedly, the highest NO concentrations were observed at sites in close proximity to traffic emissions, i.e., the adjacent-highway (8 ± 10 ppb) and near-roadway (5 ± 6 ppb) sites. The study-duration average NO concentration at the near-airport residence (1 ± 2 ppb overall and 2 ± 3 ppb during impact-sector winds) was 5-fold higher than at the regional-background site (0.2 ± 1 ppb), comparable to the urban-background site (2 ± 4 ppb), and many-fold lower than at the adjacent-highway and near-roadway sites. It is noteworthy that our study site is also farther downwind of the airport than the near-roadway regulatory sites are to traffic emission sources; thus, we likely measured a more aged plume with greater NO_2 relative to NO. See discussion in SI (Section S2.7) for other pollutants.

The study-duration average outdoor PNC as well as indoor PNC at the near-airport residence exceeded the outdoor PNC at the two UFP monitoring sites for all hours of the day (Figure 5h). The near-airport residence study-duration average concentrations were 18 000 ± 31 000 particles/cm^3 outdoors and 13 000 ± 20 000 particles/cm^3 indoors with the impact-sector averages being 38 000 ± 42 000 and 25 000 ± 27 000 particles/cm^3, respectively. In comparison, the ambient average PNC at the Chelsea site was 11 000 ± 9700 particles/cm^3 and 12 000 ± 5900 particles/cm^3 at the urban-background site. Near-airport indoor averages were comparable to the median 8000−27 000 particles/cm^3 concentrations measured on-road with a mobile lab in Boston and Chelsea[59] and to the 25 000 particles/cm^3 median concentration reported within 0−50 m of I-93 during summer; all-season median was 37 000 particles/cm^3, which was comparable to the outdoor median concentration during impact-sector winds at the near-airport residence.[43]

IMPLICATIONS

Our results show that when jet airplanes used preferred runways during impact-sector winds, particularly when such a configuration included overhead descents, outdoor and indoor PNC were remarkably elevated at our residential monitoring site ~1 km from the Logan Airport. Temporally, the highest PNC coincided with the periods of highest noise co-exposures (i.e., overhead landing flight hours). This finding is consistent with previous studies that have investigated the spatial patterns of pollutants around airports and have shown that PNC is significantly elevated downwind,[24] but especially under landing jet trajectories coinciding with the highest noise impact contours.[25] Our work underscores the need to account for both aviation-origin air pollution and noise co-exposures to avoid potential confounding of health risk associations to airport proximity.

Further, by clearly demonstrating the relationships between meteorological forcings (e.g., wind direction and wind speed) and aviation activity on UFP infiltration, our results add to the nascent body of knowledge of airport impacts on surrounding neighborhoods. These findings have implications for exposure assessment: exposure monitoring campaigns should be designed to include adequate coverage of the times of day (and times of high flight activity) with specific meteorological conditions of concern, especially wind direction. Our results also show that in the vicinity of airports, exposure to pollutants, particularly UFP and NO_2, is as significant in magnitude as that observed in the vicinity of highways. Also, we observed that indoor PNC were comparable to on- and near-highway PNC and that ambient NO_2 concentrations exceeded those observed at regulatory monitoring sites near an interstate highway and major arterial roadways. It is noteworthy that at this residence (and nearby areas),[60] PNC were highest during the evening and night-time hours (1700−2300 h), the times that people spend most of their time at home. In contrast, the lowest PNC in near-highway homes and on-road in the Boston metropolitan area occur during the late evening to overnight hours.[59,61] Compared to investigations of near-highway exposures to UFP and other traffic-related air pollutants, near-airport exposures remain essentially unaddressed in the literature.[16]

While our results provide a basis for better characterizing exposures to air pollutants of aviation origin at near-airport residences, additional work is needed to assess generalizability. For example, further work is needed to quantify the impact of housing stock characteristics (age, architectural style, and degree of sound insulation) on infiltration. Likewise, studying a greater range of behaviors that impact infiltration and indoor air quality (e.g., air conditioner use, in-home filtration, and ceiling fans) could help to identify practices that reduce indoor exposures. In addition, because we conducted our study in summer, it would be informative to repeat it in winter to quantify seasonal differences in both outdoor air quality and indoor infiltration; both are expected to differ seasonally. Similarly, because we only measured PNC infiltration, it would be useful to measure additional pollutants indoors (e.g., NO_2 and BC) to determine whether other pollutants infiltrate to the same extent as PNC. Finally, the chemical composition of aviation-related particulate air pollution at the neighborhood or community scale (i.e., few to tens of kilometers from the airport) remains unaddressed in the literature. Studies of the chemical composition of particles may shed light on the relative contributions from landings, takeoffs, idling, and taxiing at this scale and may also provide insights into mitigating these impacts (e.g., benefits derived from reducing idling times).

ASSOCIATED CONTENT

Supporting Information

The Supporting Information is available free of charge at https://pubs.acs.org/doi/10.1021/acs.est.0c01859.

Maps showing preferred runway configurations, traffic around the near-airport site and regulatory sites, details of instruments, summary of concentrations, diurnal trends for pollutants and meteorological parameters, correlations between pollutants and between pollutants and flight activity, illustration of particle infiltration and trends of the I/O ratios with respect to the temporal and meteorological parameters, and comparison of near-airport concentrations to those at regulatory sites including the diurnal patterns and their discussion and a concentration summary table (PDF)

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Notes

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Environ. Sci. Technol. XXXX, XXX, XXX−XXX