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16. Abstract

Background: Sleep disruption is linked with chronic disease, and aircraft noise can disrupt sleep. However, there are few investigations of aircraft noise and sleep in large cohorts.

Objectives: We examined associations between aircraft noise and self-reported sleep duration and quality in the Nurses' Health Study, a large prospective cohort.

Methods: Aircraft nighttime equivalent sound levels (Lnight) and day–night average sound levels (DNL) were modeled around 90 U.S. airports from 1995 to 2015 in 5-y intervals using the Aviation Environmental Design Tool and linked to geocoded participant residential addresses. Lnight exposure was dichotomized at the lowest modeled level of 45 A-weighted decibels [dB(A)] and at multiple cut points for DNL. Multiple categories of both metrics were compared with less than 45<45 dB(A). Self-reported short sleep duration (less than 7<7 h/24-h day) was ascertained in 2000, 2002, 2008, 2012, and 2014, and poor sleep quality (frequent trouble falling/staying asleep) was ascertained in 2000. We analyzed repeated sleep duration measures using generalized estimating equations and sleep quality by conditional logistic regression. We adjusted for participant-level demographics, behaviors, comorbidities, and environmental exposures (greenness and light at night) and examined effect modification.

Results: In 35,226 female nurses averaging 66.1 years of age at baseline, prevalence of short sleep duration and poor sleep quality were 29.6% and 13.1%, respectively. In multivariable models, exposure to Lnight greater than or equal to 45≥45 dB(A) was associated with 23% [95% confidence interval (CI): 7%, 40%] greater odds of short sleep duration but was not associated with poor sleep quality (9% lower odds; 95% CI: negative 30 percent−30%, 19%). Increasing categories of Lnight and DNL greater than or equal to 45≥45 dB(A) suggested an exposure–response relationship for short sleep duration. We observed higher magnitude

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Associations between Aircraft Noise Exposure and Self-Reported Sleep Duration and Quality in the United States-Based Prospective Nurses' Health Study Cohort

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Objectives: We examined associations between aircraft noise and self-reported sleep duration and quality in the Nurses' Health Study, a large prospective cohort.

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Discussion: Aircraft noise was associated with short sleep duration in female nurses, modified by individual and airport characteristics. https://doi.org/10.1289/EHP10959

Introduction

Sleep is an essential, natural process needed for healthy brain and general body functioning.[1] Disruption of sleep can cause drowsiness and poor concentration and adversely affect the metabolic, endocrine, and immune systems.[2–5] Poor sleep quality and short sleep duration (alternatively referred to as insufficient sleep or reduced total sleep time) have been associated with many adverse health outcomes,[6] including depression,[7] metabolic disorders (e.g., obesity, type 2 diabetes),[8–10] cardiovascular disease,[11] incident ulcerative colitis,[12] coronary events,[13] cancer,[14] kidney function decline,[15] mental and physical functional decline,[16–18] and mortality.[19–21]

Noise is unwanted or harmful sound,[22] where sound is defined as repetitive variations in air pressure (i.e., vibrations) sensed by the human ear. Humans recognize, evaluate, and react to environmental sounds even when asleep.[23] Noise can disrupt sleep architecture via arousals, sleep-stage changes, and awakenings.[24,25] These, in turn, increase cortical excitations, indicative of an elevated stress response to noise, which may activate the sympathetic nervous system.[24]

Aircraft noise is unique for its multispectral acoustical properties that impact the human auditory system[26] and has been shown to disrupt sleep.[24,25,27] However, most studies of aircraft noise and sleep have taken place in a small collection of homes,[28] human sleep laboratories,[25,29,30] or around one or a few airports.[28,31–37] Despite calls by researchers for more large-scale field studies,[24] only two previous studies to our knowledge have used large-scale methods across many airports.[38,39] In addition, most studies have been cross-sectional in design and therefore unable to assess associations over time. Sleep patterns may be disrupted by lifestyle factors such as shift work,[40] as well as environmental exposures that include air pollution, greenness, inopportune light at night (LAN), and noise.[41–45] But to our knowledge, there are no large-scale studies of aircraft noise that integrate and adjust for multiple environmental exposures such as greenness and LAN together, particularly in the United States,[38] where the health effects of aircraft noise have been understudied. Although a legal framework for noise abatement was established for high aircraft noise levels [i.e., ≥65 dB(A)] in the United States,[46,47] few studies have assessed potential health effects at lower thresholds in the country. Finally, there has been limited assessment of effect measure modification in previous studies to identify potential vulnerable and susceptible subpopulations.

We seek to add to existing knowledge by investigating associations between aircraft noise and sleep repeatedly self-reported in a U.S.-based prospective cohort living near 90 U.S. airports,

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controlling for individual and area-level factors. We also examine whether any relationships found are modified by individual, airport, or area characteristics.

Methods

Study Population and Airports

We used the Nurses' Health Study (NHS), an ongoing nationwide U.S. prospective cohort started in 1976, in this research. The cohort was comprised of 121,701 female registered nurses who were 30–55 years of age at initial enrollment. Although recruitment originally focused on the 11 states with the most registered nurses, participants have moved during follow-up and now live throughout the United States. Questionnaires were self-administered biennially and contained questions about incident disease, medical history, and lifestyle factors. Response rates have been $\geq 90\%$.[48,49] Home addresses indicated by the participants were geocoded every 2 y (corresponding with the biennial sequence of the NHS).

We used responses to questionnaires in survey years within the interval with aircraft noise assessment (1995–2015) that included questions about sleep—2000, 2002, 2008, 2012, and 2014—to define the study years. The year 2000, the first year for which both aircraft noise and sleep data were available, was the study baseline. To be included in this study, participants needed to have reported on sleep and have a valid residential address in the United States that could be used to successfully locate them geographically (i.e., geocoded). Residential moves among the participants during the study years were captured at the biennial assessment and aircraft noise exposure and environmental exposure estimates were updated, but participant-years were skipped when new addresses could not be geocoded. Participants were further excluded from the study if they did not live within a 22.2-mi (35.7-km) radius buffer around 1 of the 90 study airports at baseline, which represented the maximal empirical extent of aircraft noise above a day–night average sound level (DNL) of 45 decibels [A-weighted, dB(A)] surrounding any of the airports in this study. This exclusion criterion was used to limit the people unexposed to aircraft noise [here referring to those below our lowest modeled level of 45 dB (A)] to those who lived in places similar to where the exposed group lived and, therefore, reduce potential bias from unmeasured confounding factors associated with living close to an airport. Use of the 22.2-mi (35.7-km) buffers additionally helped limit exposure misclassification of participants living near airports that were not included in this study. Table S1 summarizes the number of participants excluded at each survey year for each successive criterion.

The 90 study airports were selected based on availability of model input (aircraft operations) data. They were located in 40 of 50 states plus the District of Columbia and captured 87% of all enplanements in the United States in 2010.[50]

The study protocol was approved by the institutional review board of Brigham and Women's Hospital, Boston, Massachusetts. Consent was implied through the return of the questionnaires.

Outcome Assessment

Participants self-reported the average number of hours they slept in a 24-h day in 5 survey years [2000 (baseline), 2002, 2008, 2012, and 2014]. Response options included ≤ 5, 6, 7, 8, 9, and ≥ 10 h/24-h day. Self-reported sleep duration on the 2002 questionnaire was previously shown to correlate strongly ($r_{\text{Spearman}} = 0.79$) with sleep assessed by a 1-wk sleep diary in a validation study among a subset of NHS participants.[21] The American Academy of Sleep Medicine and the Sleep Research Society reports that physiological and neurobehavioral deficits can occur and worsen over time with <7 h of sleep every 24 h.[14,51] Therefore, we defined short sleep duration as <7 h/24-h day. For the sleep quality outcome, participants were asked a question at baseline (2000) about their frequency of having difficulty falling or staying asleep during the previous 4 wk. We defined poor sleep quality as participant answers to this question on a six-point Likert scale of: "all of the time," "most of the time," or "a good bit of the time."

Exposure Assessment

Details about the generation of aircraft noise estimates are described elsewhere.[52,53] Briefly, aircraft noise estimates around all 90 study airports were modeled comprehensively using analogous input data and modeling assumptions by the U.S. Department of Transportation's John A. Volpe National Transportation Systems Center (Volpe) using the U.S. Federal Aviation Administration's (FAA's) Aviation Environmental Design Tool (AEDT; https://aedt.faa.gov/). Source data for the AEDT were aircraft operations from the Enhanced Traffic Management System, excluding helicopter operations. All annualized operations (e.g., commercial, cargo, military) were grouped by Aircraft Noise and Performance aircraft type, day or nighttime, operation airport, and stage length.

Aircraft noise level contours were modeled every 5 y from 1995 to 2015 (1995, 2000, 2005, 2010, and 2015) surrounding the 90 U.S. airports. Two noise metrics were estimated: aircraft nighttime equivalent sound level (Lnight, A-weighted) and DNL (A-weighted). Noise levels ranged from 45 to 71 dB(A). Lnight was the primary aircraft noise metric in this analysis because it captures aircraft noise occurring when people typically sleep. However, DNL incorporates nighttime estimates as well, and it is the primary metric the FAA uses to inform decision-making about aircraft noise.[54] Where Lnight assesses aircraft noise from 2200 to 0700 hours, DNL is a 24-h annualized average, capturing the average day of a year's operations with a 10 dB(A) penalty for nighttime aircraft noise from 2200 to 0700 hours, when levels of background noise tend to be lower compared with daytime. We therefore included DNL as a secondary aircraft noise exposure metric.

Spatially, each geocoded participant home address point was linked to the aircraft noise contour polygon in which it resided and assigned the respective aircraft noise level. If an address point was not within any contour, it was assigned an arbitrary value of 44 dB(A), just below the lowest modeled value of 45 dB(A), which indicated exposures estimated to be <45 dB(A), for both Lnight and DNL. DNL <45 dB(A) aligns with the recommendation and guideline for protecting health by limiting aircraft noise from the World Health Organization (WHO) Regional Office for Europe using the day–evening–night average sound level (Lden), an aircraft noise metric comparable to DNL that includes an extra penalty for evening-time aircraft noise. However, the 45 dB(A) threshold we used for Lnight is higher than the corresponding WHO guideline of limiting Lnight to <40 dB(A).[55]

Temporally, the aircraft exposure estimates for a given year were matched to current home addresses of participants of the same survey year when they coincided. When the 5-y aircraft noise estimates were not temporally coincident to a survey year, the most recent previous aircraft noise estimates were temporally matched.

Covariates and Potential Confounders

A directed acyclic graph (DAG) proposed by Billings et al. was used to guide our theoretical DAG (Figure S1) and an operational DAG (Figure S2) on the adjustment for covariates and potential confounding factors.[42,56] Based on variables identified in the DAG, we additionally determined inclusion of environmental variables in final models after evaluating correlation magnitudes among these variables and whether they changed the effect estimates by at least 10%.

Demographic factors included age (continuous), age^2 (continuous), U.S. Census region of residence (Northeast, Midwest, South, West), race (White, Black, American Indian, Asian, Hawaiian), and individual socioeconomic status (SES). Race was derived from an NHS algorithm using a 1992 question about major ancestry in combination with a 2004 question about best fitting race category and was non–time-varying. Race is a social construct, and we include it as proxy metric to aid in adjusting for socio-historical and multigenerational effects of discrimination and racism, which have been shown to affect sleep.[57] The categories we analytically used for race mirror the survey response options. Age and census region of residence were reported every 2 y. Including age^2 in statistical models adjusted for quadratic effects of age on our sleep disturbance-related outcomes. Individual SES was captured by two variables, currently living alone (yes, no) and spouse's education (<high school, high school, >high school, missing/not married). In the NHS cohort, typical measures of individual SES are deemed less applicable given that participants are or were all nurses of moderate-to-high SES; instead, living alone and lower spousal education levels, indicating less or no joint financial resources, are used as markers of lower relative individual SES among this population.[58] Spouse's education was reported in 1992 and carried forward, whereas living alone was reported in 2000, 2008, and 2012 and was carried forward as necessary. A question asked about shift work in the 1988 survey year indicated that although 49.1% of participants reporting previously working night shifts at some point in their careers, in 1996 only 2.9% of NHS participants reported having rotating night shift work in the previous 6 months. Given the low frequency of shift work 4 y prior to this study's baseline year (2000) and no further data points on shift work after 1996, shift work was not included as a potential confounder. Biannually varying postmenopausal status until 2002 (yes, no, missing) and hormone replacement therapy (never, current, former, missing) were included in a sensitivity analysis, but these individual metrics did not vary significantly, likely because 98.8% of the participants were already post menopausal at baseline (Table 1). Behaviors included smoking status (never, former, current, missing) and alcohol consumption (none, >0–4 g/d, 5–9 g/d, 10–14 g/d, 15–29 g/d, ≥ 30 g/d, missing) and were reported every 4 y and carried forward 2 y as necessary. Comorbidities included diabetes and hypertension, determined by a self-report of a current clinician diagnosis, and were reported every 2 y. Potential environmental confounders included air pollution [particulate matter with an aerodynamic diameter of ≤ 2.5 μm ($PM_{2.5}$) per cubic meter], greenness, LAN, population density, and neighborhood SES (nSES).

Air pollution, particularly $PM_{2.5}$ and nitrogen oxides, has been associated with sleep apnea,[59,60] likely through inflammatory effects on upper airway function.[42] The annual average of monthly ambient $PM_{2.5}$ (i.e., each 1-month average in the calendar year of survey year, averaged) was estimated at participant residential addresses using generalized additive mixed models developed for spatiotemporal prediction.[61] The prediction models integrate monitored data from the U.S. Environmental Protection Agency's Air Quality Monitoring System and other publicly available networks with geospatial predictors (e.g., road network data, land use, elevation). Predictive accuracy for $PM_{2.5}$ was high ($R^2 = 0.77$). $PM_{2.5}$ estimates were made through 2007; estimates from 2007 were carried forward to the remaining study years for each participant.

Greenness from natural vegetation has been linked with longer sleep duration, improved cardiovascular biomarkers, and decreased sympathetic activation and may be associated with improved mental health and healthy behaviors promoting quality sleep, such as walking.[42,62,63] However, greener areas have also been linked with more aircraft noise annoyance,[45] potentially resulting from greener areas being quieter and/or pleasing and thus modifying the experience of noise from aircraft. Greenness was estimated via the Normalized Difference Vegetation Index (NDVI), or the ratio of the difference between the near-infrared region and red reflectance (isolating visible light absorbed by chlorophyll in plants) to the sum of the two measures[64] from 30-m resolution Landsat data. We calculated annual average NDVI at a spatial resolution of 270 m annually using R (version 4.0.3; R Development Core Team) and Google Earth Engine, matching the corresponding grid cell to the home address of each study participant. We chose to use a 270-m buffer, which is roughly the size of an average city block in Manhattan, New York, because this size is more representative of the immediate greenness environment that could affect sleep.

Light is a major input into the timing of the circadian system. LAN has been documented as having adverse effects on sleep.[65–67] Assessment of outdoor LAN was described by James et al.[68] Briefly, annual average LAN was estimated using nighttime radiance units (in nanowatts per centimeter squared per steradian) at a spatial resolution of 30 arc-seconds, or $\sim 1\ km^2$. LAN estimates were developed from satellite imagery data from the U.S. Defense Meteorological Satellite Program's Operational Linescan System, managed by the National Oceanic and Atmospheric Administration's (NOAA's) Earth Observation Group.[69] LAN values were estimated for each survey year until 2010, after which estimates were carried forward.

Dense urban built environments have been linked to positive influences on health by promoting walking and other physical activity,[70,71] but urban density may come at the cost of reduced and poorer quality sleep.[72] Population density and neighborhood metrics of room crowding have previously been associated with worse sleep outcomes.[72–74] Biannually varying census tract population density (in number of people per kilometer squared) metrics were derived from the 2000 U.S. Census for the 2000 and 2002 survey years and from the 2010 U.S. Census for the 2008, 2012, and 2014 survey years.

nSES is another potential confounder of aircraft noise affecting sleep, but previous findings have been mixed. Neighborhood disadvantage has been associated with adverse sleep outcomes in some studies[75–77] but not in others.[78–80] The social environment may be more highly associated with sleep disruption than nSES, directly.[79] However, previous research has suggested that nSES can act as an upstream factor affecting downstream mediating environmental exposures influencing sleep, including aircraft noise.[42,81] The nSES metric comprised a summed z-score of SES-related census variables (e.g., race, education, income, home value, nativity, unemployment) at the level of the census tract.[82] Annual environmental/area exposures considered in statistical models ($PM_{2.5}$, NDVI, LAN, population density, and nSES) were assigned to participant home addresses every 2 y to match the current home addresses of participants given that some of them moved residences throughout the study period.

Potential Effect Modifiers

Potential effect modifiers were determined *a priori* from hypothesized influences of individual, airport, and geographic characteristics on the aircraft noise and sleep relationship. We hypothesized that the association between aircraft noise and sleep would vary by census region (Northeast, Midwest, South, West) in the United States.[83] based on potential underlying acoustical differences resulting from seasonal and weather (e.g., temperature, humidity) factors,[84,85] geographical development patterns,[86,87] housing types,[88,89] and intensity and daily temporal distribution of air services and peak hours of operation,[90]

Table 1. Age-standardized characteristics of women in the Nurses' Health Study (NHS) at baseline (2000) overall, by nighttime aircraft sound (Lnight) exposure group, and by aircraft day–night sound level (DNL) exposure group.

Characteristic	Overall $N = 35{,}381$	Lnight <45 dB(A) $N = 34{,}838$	Lnight ≥45 dB(A) $N = 543$	DNL <45 dB(A) $N = 28{,}544$	DNL 45–54 dB(A) $N = 5{,}882$	DNL 55–64 dB(A) $N = 916$	DNL ≥65 dB(A) $N = 39$
Demographics							
Age [y (mean ± SD)]	66.1 ± 7.2 y	66.1 ± 7.2 y	67.0 ± 7.1	66.1 ± 7.2	66.1 ± 7.1	66.6 ± 7.1	68.3 ± 6.5
Region of residence (%)							
Northeast	48.7	48.5	58.7	46.7	56.8	57.4	59.1
Midwest	14.8	14.8	10.9	14.8	15.5	10.3	8.7
South	19.3	19.4	16.2	19.9	16.9	18.2	9.1
West	17.2	17.3	14.2	18.6	10.9	14.1	23.0
Race (%)							
White	96.1	96.2	89.1	96.6	94.5	90.7	86.6
Black	2.4	2.3	9.8	1.9	3.8	7.4	13.4
American Indian	0.2	0.2	0.0	0.2	0.3	0.2	0.0
Asian	1.3	1.3	1.1	1.2	1.3	1.6	0.0
Hawaiian	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Currently live alone	21.6	21.6	25.6	21.4	22.4	24.0	33.0
Spouse's education (%)							
<High school	4.0	4.0	2.9	3.8	4.6	4.5	0.0
High school	26.7	26.6	32.6	26.4	27.0	33.7	28.6
>High school	44.9	45.0	36.0	45.7	42.3	36.5	47.9
Missing	24.4	24.4	28.4	24.0	26.1	25.3	23.6
Postmenopausal (%)							
Yes	98.8	98.7	99.1	98.7	98.9	99.0	100.0
No	1.2	1.2	0.9	1.2	1.1	1.0	0.0
Missing	0.1	0.1	0.0	0.1	0.0	0.0	0.0
Hormone replacement therapy (%)							
Never	23.0	22.9	28.8	22.3	26.0	27.8	14.3
Former	27.2	27.2	26.9	27.2	27.5	27.4	39.5
Current	42.9	43.0	37.0	43.9	39.0	36.1	39.6
Missing	6.9	6.9	7.4	6.7	7.6	8.7	6.6
Behaviors							
Smoking status (%)							
Never	42.9	42.9	44.7	43.2	41.6	42.9	54.0
Former	47.8	47.8	47.5	47.7	48.0	47.1	43.1
Current	9.1	9.1	7.3	8.9	10.2	9.8	2.9
Missing	0.2	0.2	0.5	0.2	0.2	0.2	0.0
Alcohol consumption							
Nonmissing [%; g/d (mean ± SD)]	91.1; 5.3 ± 9.1	91.2; 5.3 ± 9.2	89.4; 4.0 ± 7.5	91.3; 5.3 ± 9.2	90.4; 5.1 ± 9.0	90.0; 4.5 ± 8.1	85.8; 3.1 ± 5.0
Missing (%)	8.9	8.8	10.6	8.7	9.6	10.0	14.2
Comorbidities (%)							
Diabetes	6.1	6.1	7.8	6.1	6.1	8.1	5.6
Hypertension	35.1	35.0	40.6	35.0	35.1	39.2	47.7
Environmental (mean ± SD)							
NDVI greenness index	0.4 ± 0.1	0.4 ± 0.1	0.3 ± 0.1	0.4 ± 0.1	0.3 ± 0.1	0.3 ± 0.1	0.3 ± 0.1
LAN ($nW/cm^2/sr$)	38.7 ± 25.2	38.4 ± 24.9	61.5 ± 30.1	35.5 ± 22.9	51.2 ± 28.8	59.7 ± 30.3	63.1 ± 18.0
$PM_{2.5}$ ($\mu g/m^3$)	13.0 ± 2.7	13.0 ± 2.7	13.7 ± 2.5	12.9 ± 2.7	13.4 ± 2.4	13.6 ± 2.5	14.2 ± 2.2
Population density (ppl/km^2)	1,929.3 ± 3,724.5	1,888.8 ± 3,642.3	4,478.5 ± 6,728.3	1,611.1 ± 2,914.1	3,098.7 ± 5,848.2	4,173.4 ± 5,848.2	7,619.2 ± 9,457.4
nSES (z-score)	−0.4 ± 3.3	−0.4 ± 3.3	−1.0 ± 2.5	−0.4 ± 3.3	−0.4 ± 3.0	−0.9 ± 2.7	−0.8 ± 1.9
Sleep (%)							
Insufficient sleep	29.6	29.5	37.6	29.2	30.8	35.4	36.8
Poor sleep quality	13.1	13.1	12.2	13.2	12.5	13.5	21.6

Note: Environmental exposures considered included a 270-m buffer of greenness (NDVI), LAN measured in units of radiance (nanowatts per square centimeter per steradian, $nW/cm^2/sr$), annual $PM_{2.5}$ ($\mu g/m^3$) at participant residential address, census tract population density (number of people per square kilometer, ppl/km^2), and census tract socioeconomic status z-score (nSES). Missing values are excluded from calculation of descriptive statistics. The study sample was age-standardized to reflect the age distribution of the NHS cohort. LAN, light at night; NDVI, Normalized Difference Vegetation Index; nSES, neighborhood socioeconomic status; $PM_{2.5}$, particulate matter with an aerodynamic diameter of ≤ 2.5 μm; SD, standard deviation.

which were covariates for which we did not have direct measurements. We also hypothesized that living near a major cargo airport could result in a stronger effect because cargo aircraft are typically larger and older planes with less aircraft noise reduction technology and frequently operate during the night.[91] Heavier aircraft climb more slowly and subsequently generate higher noise exposure on the ground. Most of these components of cargo aircraft operations are captured in AEDT modeling. However, AEDT uses the stage length method that uses takeoff-to-landing distance as a proxy for aircraft weight,[92] which may more accurately estimate the weight of passenger aircraft than the weight of cargo aircraft. We identified the 24 largest cargo airports of the 90 study airports by total landed weight of all-cargo operations.[50] The 24 cargo airports were consistently among the top 25 in landed weight over the study period.[50] We further hypothesized that the association would be stronger for participants living near airports adjacent to a large water body (i.e., water-adjacent airports), where water and local weather could acoustically alter or enhance the experience of noise from aircraft in such areas.[93,94] Our aircraft noise metrics likely did not capture the within-year variation in weather conditions that can occur near large water bodies. There were 21 water-adjacent airports, which were determined by assessing whether any existing runway configuration at an airport allowed for an overwater approach or departure. In 2008 and 2012 most participants answered questions about hearing loss. Regardless of the cause of hearing loss, we posited that greater hearing loss would decrease the magnitude of the association between aircraft noise and short sleep duration. We were unable to investigate potential effect modification of hearing loss relative to sleep quality because data on hearing loss were not available for the relevant survey year. Owing to small numbers in some strata, hearing loss was categorized as none, mild, moderate/severe, and missing.

Statistical Analysis

We created a map of the locations of the 90 airports included in this study as points using ArcGIS (version 10.8.1; ESRI). Counts of NHS participants around each airport included at study baseline were categorized into quartiles. Regions were indicated as the four U.S. Census regions in the context of state outlines using shapefiles from the U.S. Census (https://www.census.gov/geographies/mapping-files/time-series/geo/tiger-line-file.html).

We examined associations between aircraft noise and short sleep duration longitudinally using generalized estimating equations for repeated measures, and poor sleep quality cross-sectionally at baseline using conditional logistic regression. For repeated measures, an unstructured covariance matrix was used to maximize model flexibility given that we were not limited by degrees of freedom. We used SAS (version 9.4; SAS Institute Inc.) for all statistical analyses.

A participant was defined as exposed if their respective Lnight or DNL was estimated at ≥45 dB(A). We also examined cut points of Lnight 50 dB(A) and DNL 55 and 65 dB(A). We were unable to examine Lnight cut points ≥50 dB(A) and DNL ≥65 dB(A) owing to low counts of exposed participants. To evaluate potential exposure–response with each sleep outcome, we also used mutually exclusive increasing aircraft noise exposure categories compared with <45 dB(A). We used a three-category version of Lnight [<45 (reference), 45–49, and ≥50 dB(A)] and a four-category version of DNL [<45 (reference), 45–54, 55–64, and ≥65 dB(A)]. Significance of trends (p_{Trend}) were estimated by modeling categories of aircraft noise exposures (e.g., four-category DNL) as continuous variables. We also examined continuous aircraft noise exposure levels >44 dB(A), but only as an ancillary analysis because 98.5% and 80.7% of participants were below the lowest available estimate of 45 dB(A) for Lnight and DNL, respectively.

We built five models for each outcome, starting with a crude model (model 0) adjusting only for age, age^2, and calendar period. Model 1 further adjusted for demographic factors, such as region of residence, race, living alone, and spouse's education. Models 2–4 then successively adjusted for behaviors (smoking status and alcohol consumption), comorbidities (diabetes and hypertension), and environmental factors (greenness and LAN), respectively, in which model 4 was the final, fully adjusted model. Given the high correlation between LAN and population density (Table S2), we chose to select only one of these variables for inclusion in the models. LAN was selected in favor of population density as a confounder in the final analyses owing to its possible direct effect on sleep in addition to its estimation on a relatively fine-scale, 1-km grid. Analyses included indicator variables for missing covariate data. We assessed effect measure modification hypotheses by including multiplicative interaction terms of potential effect modifiers and aircraft noise ($p_{\mathrm{Interaction}}$) in models and stratifying them by the categories of each effect modifier.

Results

Figure 1 shows the location of the 90 airports included in this study. Study population characteristics at baseline for the 35,381 participants are listed overall, by Lnight exposure, and by DNL exposure in Table 1. At baseline, the age [mean ± standard deviation (SD)] of participants was 66.1 ± 7.2 y, and only 13.0% were still working full time as nurses. Most participants lived in the Northeast (48.7%). Approximately 21.6% of the participants were living alone at baseline, and over half of the nurses' spouses had more than a high school education.

Characteristics were similar for aircraft noise exposure groups, with a few notable exceptions. Participants exposed to Lnight ≥45 dB(A) tended to live more in the Northeast (58.7%) and less in the remaining regions compared with unexposed participants [Lnight <45 dB(A)]. Compared with unexposed participants, those exposed to Lnight ≥45 dB(A) were more likely to be Black (9.8% vs. 2.3%), more often lived alone (25.6% vs. 21.6%), less frequently had a spouse with more than a high school education (50.3% vs. 59.5%), and had higher rates of diabetes (7.8% vs. 6.1%) and hypertension (40.6% vs. 35.0%). Participants with high levels of exposure to either Lnight or DNL also experienced, on average, higher LAN and lived in more densely populated census tracts compared with participants exposed to low levels of aircraft noise.

Over time, participants provided an average of 3.48 survey-years of follow-up. Participant characteristics did not differ substantially for those who only provided data at baseline ($N = 2,901$) vs. those who provided 2–5 survey years of data ($N = 32,480$; see Table S3). The participants tended to live near the same airport throughout the study period (95.0%) and to have stable aircraft DNL exposure within 10 dB(A) (90.5%).

Table S4 describes levels of exposure to aircraft Lnight and DNL overall and by potentially effect-modifying airport characteristics for those exposed to ≥45 dB(A) of aircraft noise; 98.5% and 80.7% of participants were exposed to Lnight and DNL <45 dB(A) of aircraft noise, respectively. Exposures did not vary much by region, living near a major cargo airport, nor living near a water-adjacent airport.

Table 2 shows results for the estimated association between aircraft noise and sleep, adjusted for individual factors, behaviors, comorbidities, and other environmental exposures. In longitudinal analysis, we found 34% greater odds [95% confidence interval (CI): 17%, 53%] of short sleep duration among those exposed to Lnight

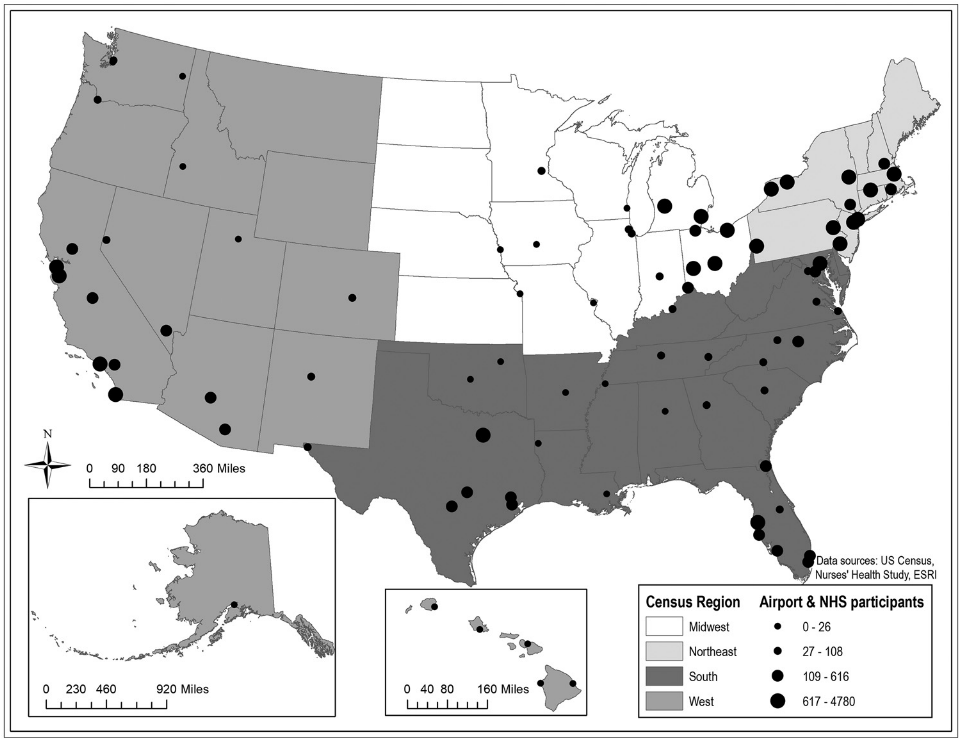

Figure 1. Map of 90 study airports across the United States symbolized by U.S. Census region and quartiles of Nurses' Health Study (NHS) participants.

$\geq$45 dB(A) compared with <45 dB(A) in the crude, age-adjusted model. Adjusting for demographics, behaviors, comorbidities, and environmental factors, the magnitude of association attenuated to 23% greater odds (95% CI: 7%, 40%) of short sleep duration among the exposed compared with the unexposed (Table 2). Using DNL as the exposure, in the fully adjusted model, we found 3% greater odds of short sleep duration for DNL $\geq$45 dB(A) relative to <45 dB(A) (95% CI: -1%, 8%). At the DNL 55-dB(A) cut point, there were stronger associations for all models, including the fully adjusted model (13% greater odds; 95% CI: 2%, 26%) (Table 2). The magnitude of the aircraft noise and short sleep duration association using a cut point of DNL 65 dB(A) was elevated (44% greater odds in the fully adjusted model) for those exposed compared with not exposed (Table 2), although the 95% CI was wide (-7%, 123%) given the small number of occurrences that any individual was an exposed case ($n = 39$; case counts by exposure category are included in Figures 2–5).

There was no evidence of a crude association between Lnight $\geq$45 dB(A) and poor sleep quality cross-sectionally (7% lower odds; 95% CI: -28%, 20%), and full adjustment for additional factors did not appreciably change the relationship (9% lower odds; 95% CI: -30%, 19%) (Table 2). Similarly, when using DNL as the exposure at 45 and 55 dB(A) cut points, we did not find evidence of associations between aircraft noise exposure and sleep quality. However, there were 101% greater odds of poor sleep quality from exposure to DNL $\geq$65 dB(A), although the 95% CI was wide and included the null (95% CI: -5%, 325%) (Table 2).

In sensitivity analyses, inclusion/exclusion of nSES, annual $PM_{2.5}$, postmenopausal status, and hormone replacement therapy, and swapping LAN for census tract population density did not change effect estimates by >10% (Table S5); therefore, these variables were not included in final models.

Using continuous versions of aircraft noise in fully adjusted models, a 5-dB(A) increase in Lnight was associated with 23% greater odds (95% CI: 6%, 44%) of short sleep duration, whereas a 5-dB(A) increase in DNL was associated with 3% greater odds (95% CI: 0%, 6%). Figures 2 and 3 show the potential exposure–response relationships between aircraft noise exposure and short sleep duration from longitudinal modeling, whereas tabular results are included in Table S6. Similar to results for dichotomized and categorical aircraft noise exposures, controlling for covariates attenuated the estimated odds ratios within ordinal categories between either Lnight or DNL and short sleep duration (Figures 2 and 3). For the crude models, results indicate the presence of an exposure–response relationship between increasing aircraft noise exposures and short sleep duration. For the fully adjusted models, results similarly suggest the presence of monotonic exposure–response relationships (Lnight $p_{\text{Trend}} < 0.01$; DNL $p_{\text{Trend}} = 0.03$).

Using continuous versions of aircraft noise >45 dB(A) in fully adjusted models, 5-dB(A) increases in Lnight was associated (Lnight: 2% greater odds; 95% CI: -23%, 34%; DNL: 2% lower odds; 95% CI: -7%, 4%) with poor sleep quality. Figures 4 and 5 (and Table S6) show that there were some indications of effects for the highest respectively exposed groups but there was no evidence of exposure–response relationships for lower exposure categories for poor sleep quality cross-sectionally (Lnight $p_{\text{Trend}} = 0.77$; DNL $p_{\text{Trend}} = 0.37$).

Results from fully adjusted longitudinal models stratified by region, major cargo airport, water-adjacent airport, and hearing loss are shown in Table 3. They suggest that the association between nighttime aircraft noise and short sleep duration differed by region of residence ($p_{\text{Interaction}} = 0.06$). The association was strongest in the West, having 83% greater odds (95% CI: 32%, 152%). Participants living near a major cargo airport had 69% greater odds (95% CI: 21%, 136%) of short sleep duration associated with nighttime aircraft noise exposure, which was higher ($p_{\text{Interaction}} = 0.09$) than those not living near a major cargo airport (16% greater odds; 95% CI: 1%, 35%). For participants living near a water-adjacent airport, the association between Lnight and short sleep duration resulted in 36% greater odds (95% CI: 12%,

Table 2. Estimated odds ratios (ORs) and 95% confidence intervals (CIs) for the relationship between aircraft nighttime sound level (Lnight), day–night sound level exposure (DNL), and short sleep duration (<7 h/24-h d) and poor sleep quality (trouble falling/staying asleep ≥ "a good bit of the time") cases at specified aircraft noise metric cut points and levels of adjustment in the Nurses' Health Study (NHS) (2000–2014).

Model	Short sleep duration $N_{obs} = 123{,}023$, $N_{participants} = 35{,}381$, $N_{cases} = 35{,}497$	Poor sleep quality $N_{obs} = 35{,}226$, $N_{participants} = 35{,}226$, $N_{cases} = 4{,}617$
Lnight ≥45 vs. <45 dB(A)		
Crude: age-adjusted	1.34 (1.17, 1.53)	0.93 (0.72, 1.20)
1: Crude+other demographics	1.27 (1.11, 1.45)	0.94 (0.71, 1.21)
2: 1 + behaviors	1.26 (1.10, 1.44)	0.94 (0.72, 1.22)
3: 2 + comorbidities	1.26 (1.10, 1.44)	0.92 (0.71, 1.20)
4: 3 + environmental	1.23 (1.07, 1.40)	0.91 (0.70, 1.19)
DNL ≥45 vs. <45 dB(A)		
Crude: age-adjusted	1.10 (1.05, 1.15)	0.96 (0.89, 1.04)
1: Crude + other demographics	1.06 (1.01, 1.10)	0.96 (0.89, 1.04)
2: 1 + behaviors	1.06 (1.01, 1.10)	0.96 (0.88, 1.04)
3: 2 + comorbidities	1.05 (1.01, 1.10)	0.96 (0.88, 1.03)
4: 3 + environmental	1.03 (0.99, 1.08)	0.94 (0.86, 1.02)
DNL ≥55 vs. <55 dB(A)		
Crude: age-adjusted	1.23 (1.11, 1.37)	1.08 (0.89, 1.30)
1: Crude + other demographics	1.17 (1.05, 1.30)	1.08 (0.89, 1.30)
2: 1 + behaviors	1.16 (1.05, 1.29)	1.08 (0.90, 1.30)
3: 2 + comorbidities	1.16 (1.04, 1.29)	1.07 (0.89, 1.29)
4: 3 + environmental	1.13 (1.02, 1.26)	1.06 (0.87, 1.28)
DNL ≥65 vs. <65 dB(A)		
Crude: age-adjusted	1.53 (0.98, 2.39)	1.94 (0.92, 4.08)
1: Crude + other demographics	1.50 (0.97, 2.31)	2.00 (0.95, 4.22)
2: 1 + behaviors	1.49 (0.96, 2.31)	2.04 (0.97, 4.30)
3: 2 + comorbidities	1.48 (0.96, 2.29)	2.04 (0.97, 4.30)
4: 3 + environmental	1.44 (0.93, 2.23)	2.01 (0.95, 4.25)

Note: Age-adjusted (age, age^2) models were sequentially further adjusted as indicated with other demographics, behaviors, comorbidities, and environmental factors. Other demographics: U.S. region of residence, race, living alone, spouse's education. Behaviors: smoking status, alcohol consumption. Comorbidities: diabetes, hypertension. Environmental: greenness (NDVI), LAN. Models of short sleep duration used generalized estimating equations to estimate odds from repeated measures in survey years 2000 (study baseline), 2002, 2008, 2012, and 2014. Conditional logistic regression models of sleep quality were used to estimate odds only for the baseline study year. dB(A), A-weighted decibel; LAN, light at night; N_{cases}, number of cases; NDVI, Normalized Difference Vegetation Index; N_{obs}, number of observations; $N_{participants}$, number of participants.

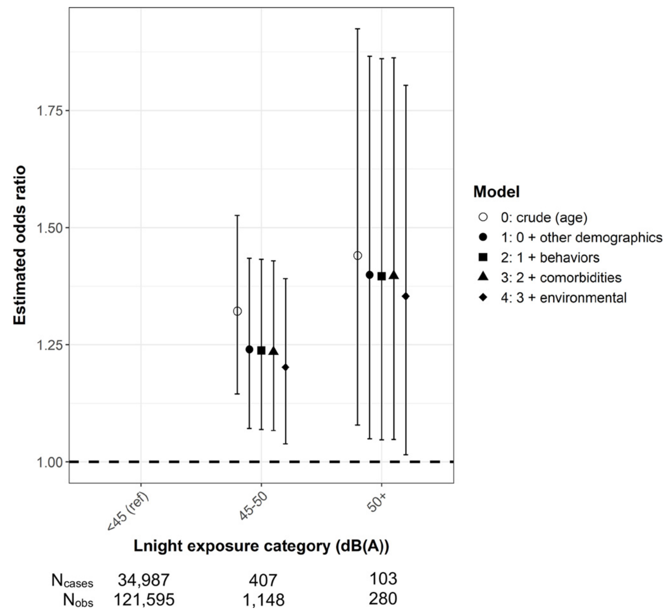

Figure 2. Odds ratio (OR) point estimates and 95% confidence intervals (CIs) investigating exposure–response relationship ($p_{Trend} < 0.01$) between categorical aircraft nighttime sound level (Lnight) exposure [<45 dB(A) (reference), 45–49 dB(A), and ≥50 dB(A)] and short sleep duration (<7 h/24-h day), using GEEs from repeated measures in survey years 2000 (study baseline), 2002, 2008, 2012, and 2014 in the Nurses' Health Study (NHS). OR and CI estimates can be found in Table S6. Models adjusted for age (age, age^2) were sequentially further adjusted for other demographics, behaviors, comorbidities, and environmental factors. Other demographics: U.S. region of residence, race, living alone, spouse's education. Behaviors: smoking status, alcohol consumption. Comorbidities: diabetes, hypertension. Environmental: greenness (NDVI), LAN. Models of short sleep duration used GEEs to estimate odds from repeated measures in survey years 2000 (study baseline), 2002, 2008, 2012, and 2014. Note: dB(A), A-weighted decibel; GEE, generalized estimating equation; LAN, light at night; N_{cases}, number of cases; NDVI, Normalized Difference Vegetation Index; N_{obs}, number of observations.

65%) compared with 11% greater odds (95% CI: −9%, 34%; $p_{Interaction} = 0.14$) for those living near a non–water-adjacent airport. For poor sleep quality in the cross-sectional analysis, there was little evidence of effect modification by region or by living near a water-adjacent airport (Table 3). However, for participants living near a major cargo airport, there were 45% greater odds (95% CI: −18%, 157%; $p_{Interaction} = 0.09$) of poor sleep quality among those exposed to nighttime aircraft noise, vs. 18% reduced odds (95% CI: −39%, 10%) for participants not living near a major cargo airport, but both intervals contained the null. Finally, for participants who reported no hearing loss, there were an estimated 50% higher odds (95% CI: 11%, 103%) of short sleep duration from exposure to nighttime aircraft noise. Estimated odds of short sleep duration were in progressively lower but also more imprecise with higher reported hearing loss [mild: 31% higher odds (95% CI: −19%, 114%); moderate/severe: 15% lower odds (95% CI: −61%, 86%)]. However, we did not find definitive evidence in the two survey years with hearing loss data available in our cohort (2008 and 2012) of an interaction between hearing loss and aircraft noise ($p_{Interaction} = 0.33$), although there was a suggested relationship.

Discussion

This study investigated the relationship between aircraft noise exposure and self-reported sleep in the NHS cohort participants living near 90 major U.S. airports. Adjusting for several individual and two environmental confounders (greenness and LAN), we found that exposure to aircraft noise was associated with short sleep duration in a repeated measures analysis, with an exposure–response relationship seemingly evident across all levels of aircraft noise included. Furthermore, we found evidence of potential effect modification by individual, area, and airport characteristics, with stronger associations with short sleep duration for participants living in the West, near major cargo airports, near water-adjacent airports, and among those reporting no hearing loss. Exposure to aircraft noise had a limited association with poor sleep quality cross-sectionally, with highly positive but uncertain associations seen only for the highest aircraft noise exposure category.

We found that short sleep duration was linked with two metrics of average annual aircraft noise exposure, Lnight and DNL. This is only partly consistent with existing literature on aircraft noise and sleep quantity. Most laboratory and residential studies have documented that aircraft noise events can shorten sleep duration,[34,95] but some found associations with longer sleep duration in conjunction with poorer sleep quality.[31,96] A national study of aircraft noise and self-reported sleep from the large-scale U.S. Behavioral Risk Factor Surveillance System surveys did not find an association between aircraft noise and short sleep duration,[39] but the study differed in numerous ways from the

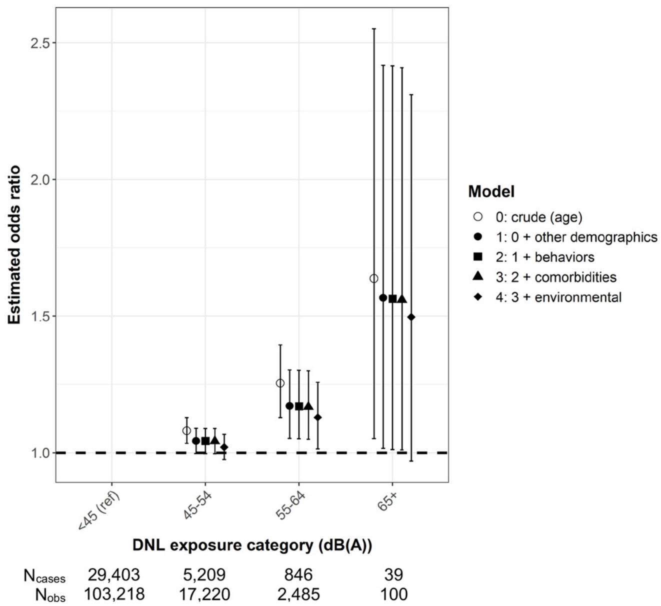

Figure 3. Odds ratio (OR) point estimates and 95% confidence intervals (CIs) investigating exposure–response relationship ($p_{\text{Trend}} = 0.03$) between categorical aircraft day–night average sound level (DNL) exposure [<45 (reference), 45–54, 55–64, and ≥65 dB(A)] and short sleep duration (<7 h/ 24-h day), using GEEs from repeated measures in survey years 2000 (study baseline), 2002, 2008, 2012, and 2014 in the Nurses' Health Study (NHS). OR and CI estimates can be found in Table S6. Models adjusted for age (age, age^2) were sequentially further adjusted for other demographics, behaviors, comorbidities, and environmental factors. Other demographics: U.S. region of residence, race, living alone, spouse's education. Behaviors: smoking status, alcohol consumption. Comorbidities: diabetes, hypertension. Environmental: greenness (NDVI), LAN. Models of short sleep duration used GEEs to estimate odds from repeated measures in survey years 2000 (study baseline), 2002, 2008, 2012, and 2014. Note: dB(A), A-weighted decibel; GEE, generalized estimating equation; LAN, light at night; N_{cases}, number of cases; NDVI, Normalized Difference Vegetation Index; N_{obs}, number of observations.

present study. Although Holt et al. established feasibility of a large-scale study of self-reported sleep, the study differed in its cross-sectional design, aircraft noise exposure modeling (Integrated Noise Model), area-level (ZIP code) exposure assignment, absence of Lnight estimates, higher DNL cut points starting at 55 dB, dissimilar reference population that included participants far from study areas and possibly near non-study airports, and potential confounders.[39] Another study around two airports in France found that a Lden 10-dB(A) increase in aircraft noise was associated with 63% greater odds (95% CI: 15%, 132%) of short total sleep time, after adjusting for individual level potential confounders.[34] However, the study did not adjust for other environmental factors, potentially confounding the relationship between aircraft noise and sleep, nor did the researchers have data available to investigate potential effect measure modification in their sample of 1,244 adults. At the area level, elevated census block group environmental noise was not associated with total hours slept in a subsample of a national cross-sectional survey of urban individuals in the United States, although it was significantly associated with other adverse sleep outcomes.[97] Although Rudolph et al. applied a novel nationwide environmental noise model, the study could not isolate the association with aircraft noise and had to rely on area-level (census block group) exposure assignment.[97] The differences in study designs used in previous research may explain their mixed results in contrast to the longitudinal design used in this study that found a strong signal between aircraft noise and short sleep duration in the NHS.

For sleep assessments, most population-based studies used relatively high cut points of the DNL aircraft noise metric, such as 50[34,62] or 55 dB,[39] although some smaller studies have assessed the influences on sleep of aircraft noise exposures as low as 30 or 35 dB.[28,31] Large-scale sleep studies investigating aircraft noise have mostly been unable to assess low potential thresholds or to incorporate nighttime specific metrics such as Lnight, particularly in the United States, with a few exceptions.[38] We found limited evidence of thresholds across the range of exposures included. When considering exposed vs. unexposed people at a 45-dB(A) cut-point, we observed a stronger association between short sleep duration and the exposure metrics for the nighttime measure than the 24-h day–night metric. Given consistent associations between sleep and a wide range of health outcomes, this finding suggests value in including nighttime aircraft noise metrics in health assessments of airport noise. Furthermore, finding evidence of an exposure–response relationship using DNL, future studies should assess a range of exposures and consider the sensitivity of conclusions about exposure categorization (Table 1).

Our findings suggest a clear, monotonic exposure–response relationship between aircraft noise and short sleep duration that is consistent with the results of smaller studies analyzing a variety of sleep parameters.[29,32,38,95] However, most of the studies documenting exposure–response relationships between transportation noise and sleep parameters often used laboratory- or home-based designs with limited adjustment for potential confounding factors that lacked generalizability to larger populations.

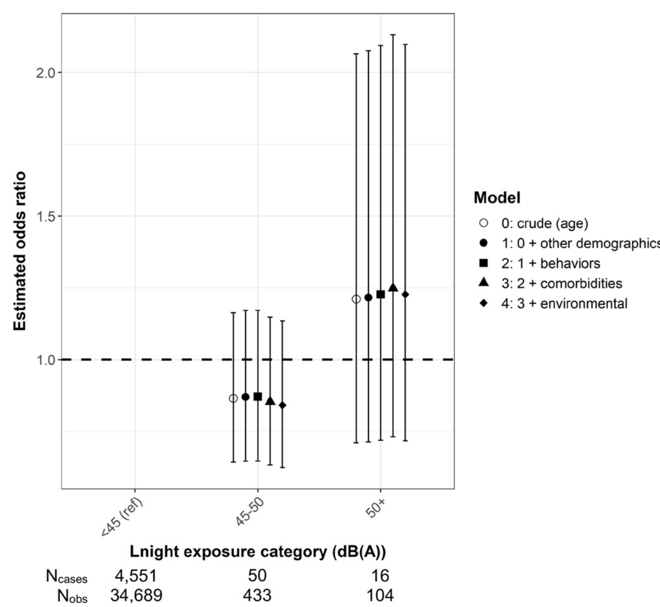

Figure 4. Odds ratio (OR) point estimates and 95% confidence intervals (CIs) investigating exposure–response relationship ($p_{\text{Trend}} = 0.37$) between categorical aircraft nighttime equivalent sound level (Lnight) exposure [<45 (reference), 45–49 dB(A), and ≥50 dB(A)] and poor sleep quality (trouble falling/staying asleep ≥ "a good bit of the time") using conditional logistic regression at study baseline (2000) in the Nurses' Health Study (NHS). OR and CI estimates can be found in Table S6. Models were adjusted for age (age, age^2) other demographics, behaviors, comorbidities, and environmental factors. Other demographics: U.S. region of residence (removed from the region-specific models), race, living alone, spouse's education. Behaviors: smoking status, alcohol consumption. Comorbidities: diabetes, hypertension. Environmental: greenness (NDVI), LAN. Conditional logistic regression models of sleep quality were used to estimate odds only for the baseline study year. Note: dB(A), A-weighted decibel; LAN, light at night; N_{cases}, number of cases; NDVI, Normalized Difference Vegetation Index; N_{obs}, number of observations.

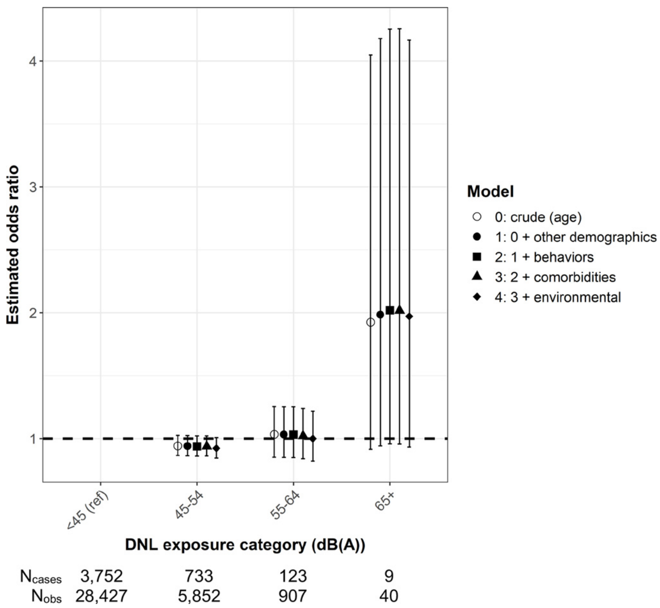

Figure 5. Odds ratio (OR) point estimates and 95% confidence intervals (CIs) investigating exposure–response relationship ($p_{\text{Trend}} = 0.37$) between categorical aircraft day–night sound level (DNL) exposure [<45 (reference), 45–54, 55–64, and ≥65 dB(A)] and poor sleep quality (trouble falling/staying asleep ≥ "a good bit of the time") using conditional logistic regression at study baseline (2000) in the Nurses' Health Study (NHS). OR and CI estimates can be found in Table S6. Models were adjusted for age (age, age^2) other demographics, behaviors, comorbidities, and environmental factors. Other demographics: U.S. region of residence (removed from the region-specific models), race, living alone, spouse's education. Behaviors: smoking status, alcohol consumption. Comorbidities: diabetes, hypertension. Environmental: greenness (NDVI), LAN. Conditional logistic regression models of sleep quality were used to estimate odds only for the baseline study year. Note: dB(A), A-weighted decibel; LAN, light at night; N_{cases}, number of cases; NDVI, Normalized Difference Vegetation Index; N_{obs}, number of observations.

An additional strength of this study was the examination of potential effect modification of the nighttime aircraft noise–sleep relationship by individual, area, and airport characteristics. Previous studies assessed potential effect modification, but in different ways. For example, one study used total sleep duration as an effect modifier and various other sleep parameters as outcomes.[38] Although greater risk for those near major cargo airports was expected, given that the aircraft noise metrics we used may not have sufficiently incorporated characteristics associated with nighttime aircraft noise dynamics, effect modification of the association with sleep had not been previously investigated. Our results suggest that participants living near major cargo airports had shorter and poorer quality sleep related to aircraft noise at night than those living near airports with less cargo activity. Evidence from Europe has shown that air cargo flights tend to use older, larger aircraft that commonly operate in the early morning hours when commercial flight activity is low.[91] In the United States, the quantity of cargo shipped by air has increased from 56.4 M ton-miles in 2003 to 76.6 M ton-miles in 2020.[98] If the trend continues, further indicated by the rapid growth in e-commerce-related cargo flight activity during the COVID-19 pandemic,[99–101] these results suggest that impacts on sleep may also increase in magnitude and for additional populations. Although our analyses incorporated noise exposure estimates that should, in theory, capture differential contributions by cargo flights, either the noise metrics we used were not well suited for characterizing sleep disturbance (e.g., differences in flight patterns or sound frequency distributions for cargo flights) or the AEDT modeling has differential error for cargo flights relative to other flights. The AEDT does not directly incorporate cargo weight because aircraft takeoff weight is proprietary in the United States; instead, the AEDT uses stage length as a proxy.[92] If true cargo weight is systematically greater than estimates from stage length formulas, such as the scenario when cargo flights have shorter stage lengths (short-haul operations) but are heavily laden, then we would expect to see differentially greater effects at airports with more cargo operations, as we indeed found.

Similarly, we found support for the hypothesis that the association between aircraft noise and short sleep duration was greater for participants living near a water-adjacent airport, but there was little evidence of a modifying effect on poor sleep quality. For water-adjacent airports, our sleep duration findings may indicate additional complexity in these acoustical environments whereby aircraft sound energy may propagate more easily over water,[93,94] differentially influencing sleep. The AEDT assumes a soft ground surface (e.g., grass) near the sound receiver. Reflections off hard ground, such as pavement, or water generally cause higher sound levels. Under certain conditions, sound propagating over water can be channeled by the reflection off the surface and then refracted downward owing to cool air being immediately above the surface with warmer air above that.

It is not clear why aircraft noise seems to vary in its association with short sleep duration by region of the country, although it potentially relates to aspects of the surrounding environment and climate zones, as well as nationally and internationally influenced flight operations scheduling. Additional research is needed to disentangle potential place-based aircraft sound propagation mechanisms, such as varied surfaces (hard vs. grassy surfaces), local weather, seasonality, the influence of atmospheric conditions during overflights,[102] and housing materials and types. Unique regional climates have been associated with patterns in heating/cooling system types and related behaviors (e.g., opening windows) to achieve thermal comfort in bedrooms while sleeping,[103] which may further modify the effects of aircraft noise on sleep. Daily flight scheduling can be a function of time zones, levels of business activity, airline hub and spoke networks, and markets served (e.g., geographic proximity to national and international destinations).[104]

There are age-related changes in hearing as people age,[105] but we found a trend suggestive of a lower magnitude relationship between aircraft noise and short sleep duration with greater hearing loss independent of age. However, this study was underpowered to robustly investigate potential effect modification by hearing loss. In the literature, hearing loss is usually mentioned as a physiological outcome of aircraft noise,[55,106] used as an exclusion criterion,[30,96] or, as in one study, considered as a confounder.[35] However, we did not find sleep studies that assessed potential effect modification by hearing loss.

We did not find associations with sleep quality in most analyses. However, the NHS surveys did not capture longitudinal self-assessments of multiple dimensions of sleep quality. Thus, we were only able to use participant answers to one question in 2000, in which only 66 of the 541 participants who were exposed to Lnight ≥45 dB(A) reported poor sleep quality. Despite limited power, we cannot rule out a potential association with poor sleep quality at very high levels of DNL exposure. Other predominantly smaller-scale studies conducted in human sleep laboratories or participant homes have found deleterious effects of aircraft noise on sleep quality.[31,35]

There were several limitations of our study. Our study population did not include males, younger individuals, or many individuals from underrepresented groups owing to the construction of the original cohort. Noise sensitivity and annoyance, which may influence the effects of aircraft noise on sleep, were not

Table 3. Estimated odds ratios (ORs) and 95% confidence intervals (CIs) for potential effect measure modifiers of the relationship between aircraft nighttime sound level [Lnight ≥45 dB(A)] and short sleep duration (<7 h/24-h day) and poor sleep quality (trouble falling/staying asleep ≥ "a good bit of the time") cases from fully adjusted models (model 4) in the Nurses' Health Study (NHS) (2000–2014).

	Short sleep duration $N_{obs}=123{,}023$, $N_{participants}=35{,}381$, $N_{cases}=35{,}497$			Poor sleep quality $N_{obs}=35{,}226$, $N_{participants}=35{,}226$, $N_{cases}=4{,}617$		
Potential effect modifier	N_{obs} (cases)	OR (95% CI)	p-Value	N_{obs} (cases)	OR (95% CI)	p-Value
Region			0.06			0.90
Northeast	59,358 (18,278)	1.15 (0.96, 1.37)		17,132 (2,251)	0.88 (0.62, 1.25)	
Midwest	18,720 (4,831)	1.33 (0.90, 1.96)		5,211 (641)	1.08 (0.51, 2.31)	
South	23,564 (6,533)	1.08 (0.77, 1.51)		6,803 (975)	1.05 (0.57, 1.90)	
West	21,381 (5,855)	1.83 (1.32, 2.52)		6,080 (750)	0.77 (0.37, 1.60)	
Water-adjacent airport			0.14			0.99
No	76,071 (20,781)	1.11 (0.91, 1.34)		21,401 (2,761)	0.91 (0.63, 1.31)	
Yes	46,952 (14,716)	1.36 (1.12, 1.65)		13,749 (1,856)	0.95 (0.65, 1.38)	
Major cargo airport			0.09			0.09
No	96,445 (27,760)	1.16 (1.01, 1.35)		27,431 (3,613)	0.82 (0.61, 1.10)	
Yes	26,578 (7,737)	1.69 (1.21, 2.36)		7,795 (1,004)	1.45 (0.82, 2.57)	
Hearing loss[a]			0.33		NA	
None	30,422 (8,974)	1.50 (1.11, 2.03)				
Mild	15,939 (4,899)	1.31 (0.81, 2.14)				
Moderate/severe	9,379 (2,711)	0.85 (0.39, 1.86)				

Note: Models were adjusted for age (age, age^2), other demographics, behaviors, comorbidities, and environmental factors. Other demographics: U.S. region of residence (removed from the region-specific models), race, living alone, spouse's education. Behaviors: smoking status, alcohol consumption. Comorbidities: diabetes, hypertension. Environmental: greenness (NDVI), LAN. Models of short sleep duration used generalized estimating equations to estimate odds from repeated measures in survey years 2000 (study baseline), 2002, 2008, 2012, and 2014. Conditional logistic regression models of sleep quality were used to estimate odds only for the baseline study year. dB(A), A-weighted decibel; LAN, light at night; N_{cases}, number of cases; NDVI, Normalized Difference Vegetation Index; N_{obs}, number of observations; $N_{participants}$, number of participants.
[a]Hearing loss was assessed and analyzed only for short sleep duration in 2008 and 2012. For potential effect modification by hearing loss: $N_{obs}=55{,}740$, $N_{participants}=25{,}627$, and $N_{cases}=16{,}584$.

included because they were not measured in the cohort. In addition, there may be residual confounding by weather conditions, such as temperature, which independently affects sleep.[74,83] Furthermore, we could not adjust for sleep medication use, which might be a potential confounder. Sleep was subjectively self-reported, and it has been shown that subjective reporting of sleep duration may underestimate objective measures, although it can depend on how the question is asked. Only a single question about sleep quality was available on one biennial questionnaire. Although aircraft noise estimates used were average sound energies over time, single-event exposures of individual flights [e.g., maximum sound level (Lmax) or sound exposure level (SEL)] may be more relevant to sleep, yet such metrics are infrequently used.[55] We could not capture within-5-y variations in noise exposure. Aircraft noise estimates were outside at home addresses, not in the actual bedrooms in which participants presumably slept, and they did not include noise from ground operations in and around airports. Although important housing and indoor home environments were not able to be captured, including sound insulation, window opening/closing or air conditioner use,[107] address-level exposure assignment for a large-scale study was a novel contribution to aircraft noise and sleep research. Other sources of environmental noise (e.g., natural, community, road, rail) were not captured directly, although they were likely partly captured by environmental confounders included. In the present study, LAN likely acted as a proxy for population density and sources of noise (e.g., road) and light happening more frequently in denser (e.g., urban) areas. The cohort and study population were not randomly distributed, given that participants were originally recruited from 11 U.S. states in 1976. For example, the Boston area, and the Northeast more broadly, was initially overrepresented. However, few nationwide populations have been followed with repeated sleep and aircraft noise measurements over time. Similarly, airports were not randomly selected into this study, but were included where operations data were available. However, the respective geographic coverage of both participants and airports were still wide over the study period, and the airports included in the study captured the vast majority of enplanements annually in the United States.[50] We did not directly account for noise reduction policies of individual airports. However, this would be indirectly reflected in the aircraft noise estimates. In addition, although the Aviation Safety and Noise Abatement Act of 1976 and the Airport Noise and Capacity Act of 1990 established a legal framework for abatement corresponding to a threshold of DNL 65 dB,[46,47] "airport sponsors have limited proprietary authority to restrict access as a means of reducing aircraft noise impacts" to local communities,[108] and, in practice, this authority is rarely exercised. There are currently no state or federal policies directly limiting aircraft noise in the United States.[46]

Conclusions

The increasing recognition of the importance of adequate sleep for maintaining health and optimal daytime functioning has spurred research aimed at identifying modifiable factors for improving sleep duration and quality. Environmental risk factors—including noise pollution—represent targets for improving sleep health that have been underinvestigated. Estimated at participant's home addresses, multiple metrics of aircraft noise were associated with self-reported short sleep duration even after adjustment for environmental characteristics, including greenness and LAN. Short sleep duration was associated with both Lnight and DNL, and the Lnight association varied by individual, area, and airport characteristics, including region, living near a major cargo airport, living near a water-adjacent airport, and by self-reported level of hearing loss. We found evidence for adverse effects on sleep at exposures as low as DNL 45 dB(A), the lowest modeled noise level, and evidence further showed an exposure–response relationship between aircraft noise and short sleep duration. There was little evidence that aircraft noise was associated with sleep quality as assessed by questionnaire at study baseline across most levels of aircraft noise exposure.

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