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Aircraft noise and environmental equity in Montréal: A comparison of noise indicators and an analysis of the impacts of COVID-19

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ABSTRACT

From an environmental equity perspective, the aim of this paper is twofold. First, we want to verify to what extent vulnerable population groups resided in areas exposed to high levels of aircraft noise before and during the COVID-19 pandemic (2019 and 2020) in the Montréal census metropolitan area. Second, we want to identify whether the use of an aircraft noise indicator rather than another generates significant variations in the results and consequently in terms of affected areas and populations.

With the IMPACT web-application, we model aircraft noise contours from three cumulative ($L_{den}$, $L_{dn}$, $L_{aeq,24h}$) and a single-event ($L_{Amax}$) metrics. The model’s input data are retrieved by a website for flight tracking. Next, four variables are extracted from the 2016 Statistics Canada census at a fine scale level (dissemination areas): that is, the percentages of low-income individuals, visible minorities, children under 15 years old, and individuals aged 65 and over.

The results show a significant drop in population exposed to aircraft noise in 2020 compared to 2019. In addition, the estimates of populations impacted by aircraft noise differ from one indicator to the next. The logistic regression models indicate that the inequities are inconsistent between cumulative and single-event metrics.

1. Introduction

Aircraft noise is considered to be the most annoying of disturbances compared to road and railway noises (Babisch et al., 2009; Miedema and Oudshoorn, 2001). Its impacts on health are now well known: psychological disturbance and stress (Stansfeld and Matheson, 2003), sleep disorders (Basner et al., 2017), increased cognitive difficulties (Clark et al., 2012), and even risks of arterial hypertension and cardiovascular diseases (Correia et al., 2013). Some population groups are more vulnerable to noise. A chronic exposure to high levels of aircraft noise can affect children’s cognitive development (e.g., learning difficulties, reduced cognitive functions, and concentration difficulties) (Basner et al., 2017). Seniors are more sensitive to the effects of noise in their residential environment due to their physiological characteristics, but also because of more reduced mobility and more time spent at home (Li et al., 2021; Muzet, 2007). Thus, when they live in spaces where noise levels are high, they are at greater risk of developing cardiovascular diseases (Correia et al., 2013; Van Kamp and Davies, 2013).

These past years, the continuous increase in air traffic and the ensuing consequences on health and quality of life in terms of
pollutant emissions generate serious concerns expressed by residents living close to airports (Baudin et al., 2018; Van den Berg et al., 2015). However, at the close of 2019, the situation radically changed with the COVID-19 global pandemic having a huge impact on air travel (Suau-Sanchez et al., 2020). Lockdown measures, border shutdowns, and travel restrictions implemented in many countries to curb the spread of the virus interrupted travel, entailing a drastic decrease of air passenger traffic. In 2020, the regular global passenger traffic (international and national) decreased by 60% compared to 2019 (ICAO, 2021). Apart from freight air traffic which remained relatively stable, the sudden drop in air passenger traffic comes with a reduction of noise emissions in the vicinity of major airports. For example, in France, during the first lockdown (March to May 2020), the reduction in noise levels around the Paris-Charles-de-Gaulle airport reached an average of 21.8 dB(A)$_{L_{den}}$ and 24.4 dB(A)$_{L_{dn}}$ compared to the usual situation (Bruitparif, 2020). Also, during some weeks, the decrease in noise related to the reduction of air traffic reached 30 dB(A)$_{L_{den}}$ in the vicinity of major airports in the Île-de-France region (Paris-Charles-de-Gaulle, Paris-Orly, and Paris-Le Bourget).

Due to the exceptional COVID-19 context, this study focuses on the representation of vulnerable groups affected by aircraft noise in their residential environment before and during the COVID-19 pandemic (2019 and 2020) in the Montréal census metropolitan area (CMA). This research will also emphasize the use of numerous noise indicators to characterize aircraft noise.

2. Literature review

2.1. Environmental equity and aircraft noise

The concept of environmental equity is based on three dimensions of justice (Schlosberg, 2007; Walker, 2012): procedural justice referring to the role of various groups composing a society in the decisional processes; recognition justice assuming respect for these
groups and refusing contempt for certain social categories; distributional justice or environmental equity which focuses on the distribution or sharing of beneficial elements (resources) and negative elements (sources of risk) in relation to the location of specific population groups (defined according to age, income, or ethnic belonging). In this study, we focus on this last dimension.

Among the work conducted on aircraft noise, transversal studies have shown that low-income households and certain ethnocultural communities (Hispanic, Black, or Asian) are subject to disproportionate exposure to high levels of aircraft noise in their residential area. This was shown in Saint-Louis (United States) (Most et al., 2004), Phoenix (United States) (Sobotta et al., 2007), Calgary and Vancouver (Canada) (Audrin et al., 2019) and across the United States (US) (Collins et al., 2020). For example, Sobotta et al. (2007) find that on average in Phoenix, the percentage of Hispanic households increases the probability of being exposed to high levels of aircraft noise by 25%. The results of the study by Audrin et al. (2019) show that the greater the increase in the percentage of low-income persons in an area, the higher the probability of being located in a Noise Exposure Forecast (NEF 25 or 30) zone in Calgary and Vancouver. Recently, Collins et al. (2020) performed a US national-level analysis of residential noise from road and air transport. Examining exposure to aircraft and road noise separately, the results showed a slight association between high level of aircraft noise and census tracts characterized by lower socioeconomic status. In other words, neighbourhoods with a greater proportion of ethnic minorities (African-American, Hispanic, Asian and Pacific Islander) and with higher deprivation and proportion of renters are disproportionately exposed to aircraft noise ($L_{Aeq,24h}$ values greater than or equal to 35 dB(A)). A reverse trend was found in neighbourhoods with a high concentration of white residents. Furthermore, among the rare longitudinal studies, that of Ogneva-Himmelberger and Cooperman (2010) in Boston (United States) reveals that there is an overrepresentation of low-income individuals and Hispanic populations in zones where the levels of aircraft noise are high compared to zones with noise levels that are lower, as much in 1990 as in 2000. In other words, despite the decrease of aircraft noise levels (i.e., size of noise contours) in the vicinity of the Boston Logan International Airport between 1990 and 2000, there was no considerable change in environmental equity patterns.

Paradoxically, other studies found contradictory results that can be explained by methodological choices, and specific geographical, historical, and regulatory contexts. For example, Most et al. (2004) analyzed the impacts of spatial scale and population assignment choices regarding exposure to high levels of aircraft noise at the Saint-Louis-Lambert airport (United States). The authors concluded that results vary significantly depending on methodological choices, which may lead to different findings during a diagnosis of environmental equity. In the Rijnmond region in the Netherlands, Kruize et al. (2007) conclude that wealthier households are overexposed to aircraft noise. Historically in this region, wealthier households reside in the suburbs of Rotterdam, in rural areas, where Rotterdam-La Haye is located. Similar observations were made in Toronto and Montréal, in Canada. A recent study found that low-income populations are slightly more advantaged in terms of exposure to aircraft noise in the metropolitan areas of Toronto and Montréal (Audrin et al., 2019). This situation can be explained by the specific residential geography of these two cities where there are mostly wealthier sectors close to the two airports in the neighbouring suburbs, whereas the mostly disadvantaged populations are concentrated in certain central neighbourhoods.

In contrast, despite their physiologically vulnerable to aircraft noise (Bassner et al., 2017; Kaltenbach et al., 2008), seniors as well as children were little accounted for in studies on environmental equity related to aircraft noise in their residential neighbourhood. The study by Audrin et al. (2019) considers these two population groups and covers the four largest Canadian cities (Toronto, Montréal, Vancouver, and Calgary). The study confirms that there is no inequity in exposure to aircraft noise for seniors, whereas there is a slight inequity for children under 15 years of age in Calgary and Toronto. Furthermore, a US national-level study (Collins et al., 2020) referred to these two population groups and the results confirmed that there is no environmental inequity in relation to aircraft noise exposure.

In summary, the equity diagnoses for exposure to aircraft noise vary from one city to the next and from one population group to the next. However, few longitudinal studies have shown an interest in this issue. It is therefore difficult to evaluate if the equity diagnosis varies from one year to the next. From a methodological point of view, there seems to be a lack of attention to the issue of choice of metrics to measure aircraft noise. Yet, in the literature on environmental equity, it has been shown that results can vary significantly depending on methodological choices (Most et al., 2004).

2.2. Measurement of aircraft noise: A brief review

The noise generated by aircraft is the main annoyance experienced by those living near airports. Controlling noise disturbances in the vicinity of airports is therefore a major issue, but it is a complex one that includes different areas of jurisdiction.

The balanced approach to noise management developed by ICAO (2001) (International Civil Aviation Organization) is an approach that consists of taking into account various possible measures to reduce aircraft noise disturbances. The principle of the balanced approach is to address issues of noise disturbances generated by aircraft for surrounding neighbourhoods, then analyze the various means available for abatement. Four major means are used: reduction of noise at source by using quieter aircraft, land-use planning and management to limit exposure to noise for populations, use of noise abatement operational procedures (e.g., specific procedures for take-off and landing), and operating restrictions (e.g., specific restrictions for certain types of aircraft or during certain time periods of a day cycle, especially at night).

For research purposes, noise contour maps are widely used to evaluate exposure to aircraft noise for populations. They provide information on the levels of aircraft noise concentration. A noise contour is a line on a map that represents equal levels of noise exposure. Each noise contour is associated to a level of noise measured in A-weighted decibels (dB(A)) to reflect the sensitivity of the human ear’s perception to certain frequencies.

Also, the World Health Organization (WHO) recommends an average exposure value to aircraft noise in residential areas of 45 dB (A) $L_{den}$ and 40 dB(A) $L_n$ to limit the adverse effects on health (WHO, 2018). However, note that the recent systematic review on aircraft
noise (Gjestland, 2018) recommends an exposure limit of 53 dB(A) \( L_{\text{den}} \). Two main types of metrics are used for the evaluation of impact of aircraft noise and the development of maps or boundaries of areas of exposure to aircraft noise: cumulative noise metrics and single-event metrics.

### 2.2.1. Cumulative noise metrics

Cumulative noise metrics take into account the cumulative noise exposure over a specified time limit (daytime, nighttime, full day). Most impact studies use them since they account for noise levels depending on the day; they are largely used in regulation since they measure average noise in the vicinity of an airport over a specific period. The development of these metrics is done with mathematical modelling. In this research, we will use three cumulative noise metrics commonly used outside Canada, that is \( L_{\text{Aeq,T}} \), \( L_{\text{den}} \) and DNL or \( L_{\text{dn}} \), whose formulas are indicated in the supplementary material (A1).

The \( L_{\text{Aeq,T}} \) metric (A-weighted equivalent continuous sound level) corresponds to the level of continuous noise equivalent weighted A. This metric cumulates all noise variations over a time period \( T \) expressed by dB(A). If solely the passing of an aircraft is of interest to us, the equivalent level will be measured according to the aircraft’s passing. This metric has traditionally been used by the WHO (Berglund et al., 1999) and it recommends, regardless of the source of noise, a noise exposure threshold not to exceed 55 dB(A) \( L_{\text{Aeq,24h}} \) whereas in England, the chosen value is 57 dB(A) \( L_{\text{Aeq,16h}} \) (CAA, 2020).

Metric \( L_{\text{den}} \) (Day-night average noise level) is sound level metric cumulated over a period of 24 h. It is calculated on the basis of equivalent values of continuous noise measured during three periods: day, evening, and night. Penalties are applied according to the time of day (compared to level during the day with the addition of 5 dB(A) in the evening, and 10 dB(A) at night). This metric is commonly used because it is representative of the disturbance experienced by the noise. For example, in Europe, for environmental noise evaluation and management, the European Environmental Noise Directive 2002/49/EC (European parliament, 2002) mandates the use of \( L_{\text{den}} \) for the development of strategic noise maps (SNM), allowing to evaluate population noise exposure generated by transport. In France, to counter aircraft noise pollution, two types of maps use \( L_{\text{den}} \). Noise disturbance plans (NDP) define zones where those residing close to airports may request assistance for soundproofing their residence, and noise exposure plans (NEP) regulate land use around airports. For NEPs and NDPS, many noise zones are generally delimited by noise contours with intervals of 5 dB(A) starting from \( L_{\text{den}} \) 50 (low noise exposure) to \( L_{\text{den}} \) 70 (very loud noise exposure).

DNL or \( L_{\text{dn}} \) (Day-night average sound level) is like \( L_{\text{den}} \). It represents total accumulation of all sound energy uniformly applied over a 24-hour period with a penalty of 10 dB(A) for the night period (10 p.m. to 7 a.m.) This type of metric is used, among others, in the United States to analyze aircraft noise exposure close to airports. The Federal Aviation Administration (FAA) has established a threshold of 65 dB(A) \( L_{\text{dn}} \) above which aircraft noise has a significant impact on the population and becomes incompatible with residential occupancy (Federal Aviation Administration, 1983).

### 2.2.2. Single-event metrics

Event metrics refer to “peak noise levels”. A peak noise can be defined as noise emergence in relation to background noise. This indicator allows to better account for single noise events that present large fluctuations over time, which is particularly relevant for aircraft noise. However, they provide no information on cumulative exposure to noise. Event metrics are mainly used in studies about the effects on sleep and short-term exposure to aircraft noise (Basner and McGuire, 2018; Quehl et al., 2017). The \( L_{\text{Amax}} \) (maximum A-weighted sound pressure level) metric corresponds to the maximum intensity associated to passing aircraft. In other words, it can be defined as the highest \( L_{\text{Aeq,1sec}} \) value (Dekoninck, 2020). Other measurements refer to single-events, such as EPNL (Effective Perceived Noise Level), SEL (Sound Exposure Level), and NAT (Noise Event Above a Threshold). Note that EPNL and SEL metrics measure total sound exposure during a single aircraft event, while \( L_{\text{Amax}} \) and NAT metrics measure the maximum level of noise. For example, NA65 corresponds to the number of events whose maximum \( L_{\text{Amax}} \) level exceeds 65 dB(A). NAT is relatively easy to interpret and is beginning to be used in different countries (Zaporozhets, 2016). However, in this study, we will only consider the \( L_{\text{Amax}} \) metric.

### 2.2.3. Aircraft noise measurements in environmental equity studies

To our knowledge, all studies on environmental equity use cumulative noise indicators for exposure to aircraft noise. More specifically, most studies use the reference indicator of the case study country: \( L_{\text{den}} \) in Europe (Kruize et al., 2007), \( L_{\text{dn}} \) and \( L_{\text{Aeq,24h}} \) in the United States (Sobotta et al., 2007; Ogneva-Himmelberger & Cooperman, 2010; Collins et al. 2020). In Canada, NEF was used to measure reference noise (Audrin et al., 2021; Audrin et al., 2019). It is important to remember that the NEF index will not be considered in this study for two main reasons. On the one hand, the parameters to measure aircraft noise have not been updated since the 1970’s (TRAN, 2019). On the other hand, our noise model used in this research does not generate this type of indicator.

In the literature on environmental equity, there is a lack of attention to the use of many indicators to measure aircraft noise. However, in the area of health, a notice from Montréal’s department of public health (Canada) used various metrics to evaluate if the noise levels associated with aircraft movement in 2009 at Montréal–Trudeau International Airport exceeded the levels suggested by the WHO, which is 55 dB(A) (Smargiassi et al., 2014). This notice compares NEF, \( L_{\text{dn}} \), \( L_{\text{den}} \), and the criterion of additional awakening induced by nighttime noise from air traffic at Leipzig/Halle airport (Basner et al., 2006), which is calculated based on \( L_{\text{Amax}} \) noise levels. It was demonstrated that the choice of the aircraft noise indicator can lead to different results in the estimations of housing units exposed to high levels of aircraft noise.

### 3. Research objectives

In the literature on environmental equity related to aircraft noise, there seems to be a consensus around the fact that results vary...
from one geographical context to another and according to population groups considered. However, very few studies examine the distribution of aircraft noise disturbances multiple times in a given period of time, which may have consequences on equity patterns. Also, from a methodological point of view, there is little research exploring the impact of choice of aircraft noise indicators used on the results of the environmental equity diagnostic. Thus, it is appropriate to verify to what extent vulnerable population groups studied – low-income individuals, visible minorities, individuals 65 years and over, and children under 15 years old – reside in areas exposed to high levels of aircraft noise before and during the COVID-19 pandemic in the Montréal census metropolitan area, and whether the use of an aircraft noise indicator rather than another generates significant variations in the results and consequently in terms of affected areas and populations.

Four types of indicators – three cumulative noise metrics and one single-event metric – calculated before and during the COVID-19 pandemic will be mobilized to answer this question. Based on various aircraft noise metrics, the objective is to verify if the situation in terms of environmental equity for these four vulnerable groups deteriorated or improved between 2019 and 2020 in the Montréal metropolitan area.

4. Methodology

4.1. Study area: Montreal census metropolitan area (CMA)

The territory under study is the second largest census metropolitan area (CMA) in Canada, namely, Montréal with a population of

Fig. 1. Study area.
4.10 million in 2016. The proportions of young people under 15 years old, individuals aged 65 and over, low-income individuals, and visible minorities in the CMA are respectively 16.9%, 16.4%, 11.7%, and 22.1%, compared to 11.2%, 16.9%, 14.2%, and 22.3% for Canada as a whole (Statistics Canada, 2016). The CMA of Montréal includes the Montréal-Trudeau International Airport (ICAO code: CYUL), which is the fourth largest airport in Canada after Toronto Pearson International Airport (CYYZ), Vancouver (CYVR), and Calgary (CYLC) International Airports. In 2018, 241,442 aircraft movements (Statistics Canada, 2018a) were registered, and approximately 18.8 million passengers (Statistics Canada, 2018b) were in transit at Montréal-Trudeau International Airport. It is located southwest on the Island of Montréal in the municipality of Dorval (Fig. 1). The airport’s bordering neighbourhoods are characterized by both wealthy residential areas located in the southern sector of Dorval and in the east part of the municipality of Pointe-Claire, but also by vast industrial zones to the north and to the east. It must be noted that the Saint-Hubert (CYHY) and Montréal-Mirabel (CYMX) airports, respectively located southeast and north of the metropolitan area (Fig. 1), are not considered in this study for two reasons. On the one hand, the Saint-Hubert regional airport accommodates mainly small aircraft used for flight schools. On the other hand, the Montréal-Mirabel airport, mainly used for cargo transport, is located in a rural zone where few people live close to this airport (Fig. 1).

4.2. Data and noise indicator

4.2.1. Population groups and scale of analysis

Similar to recent work on environmental equity in Montréal (Apparicio et al., 2016; Audrin et al., 2019; Carrier et al., 2016; Delaunay et al., 2019; Pham et al., 2012; Potvin et al., 2019), four population groups are considered in this study: low-income individuals (before tax), visible minorities (see definitions in supplementary materials, A2), children under 15 years old, and individuals aged 65 and over. The first two groups are chosen considering their socioeconomic vulnerability, and the last two for their physiological vulnerability to aircraft noise.

The numbers and percentages for these four groups are taken from the Statistics Canada 2016 census for the dissemination area (DA). A DA, comprised of many dissemination blocks (DB), includes a population of 400–700, and represents the most specific spatial breakdown for which demographic and socioeconomic data are available.

An analysis at a specific spatial scale is necessary (Chakraborty et al., 2011; Most et al., 2004) to provide a precise environmental equity diagnostic. We therefore favour the dissemination block (DB), which is the smallest geographic region for which the population and dwelling counts are available (Statistics Canada, 2016). As defined by Statistics Canada (2016), a dissemination block corresponds to a city block delimited on all sides by roads and/or boundaries of standard geographic areas (e.g., census subdivisions that have the same boundaries as municipalities and cities). However, for this breakdown, Statistics Canada only provides three variables: total population, the number of housing units, and households. To solve this problem, many authors (Carrier et al., 2016; Pham et al., 2012) propose to simply estimate the numbers in a group that is part of the block, based on the following equation:

\[ t_{ij} = t_a \frac{T_j}{T_a} \]

where \( t_{ij} \) corresponds to the group \( i \) population estimate (low-income individuals, for example) in block \( j \), \( t_a \) is population group \( i \) in dissemination area \( a \), and \( T_j \) and \( T_a \) are respectively total population in the block and in the dissemination area.

4.2.2. Aircraft noise modelling in the IMPACT web application

To predict the levels of aircraft noise at the Montréal-Trudeau International Airport, we use the web application IMPACT (Integrated Aircraft Noise and Emission Modelling Platform) version 3.36.A (EUROCONTROL, 2020) as a noise model. The model’s input data are retrieved by a website for flight tracking (Flightradar24). The three following subsections present the IMPACT noise model, input data, and collection procedure.

4.2.2.1. Aircraft noise models. According to Filippone (2014), two categories of noise prediction methods are used: 1) theoretical methods based on physical noise production and dissemination models providing very specific results, but whose input parameters are numerous and require much time for calculation; 2) best practice methods based mainly on empirical models, measurement databases and sub modes contributing to simpler and faster predictions. This last category is widely used by various civil aviation organizations (e.g., AEDT in the US, ANCON model in the UK, and ECAC Doc.29 model in the EU). Our research is based on this second type of method.

The IMPACT version 3.36.A web application by EUROCONTROL (European Organization for the Safety of Air Navigation) is based on a harmonized calculation method for the ICAO and the European Civil Aviation Conference (ECAC) to generate mapping of aircraft noise contours in the vicinity of civil airports (ICAO, 2018; ECAC, 2016). IMPACT is a best-practice segmentation aircraft noise prediction model that enables calculation of noise levels and contours around airports during a given period of time by superposing the effects of single flight events (departures and arrivals). A further description can be found in the supplementary material (A3). This technique is described in detail in the ECAC Doc.29 vol 2 (ECAC, 2016) and Doc 9911 of the ICAO, 2018.

IMPACT can do calculations for many noise metrics during a specified time period, for example \( L_{Aeq,24h} \), \( L_{den} \), \( L_{den} \), and \( L_{Amax} \). Note that in this study, \( L_{Amax} \) noise contours are calculated based on daily maximum values of each flight event recorded during a given day. When the model is released, IMPACT delivers noise contours in KML or shapefile formats that can be downloaded to be processed at a later date in a geographical information system (GIS).
4.2.2.2. **Airport data collection and processing.** In this study, we target the month of August as the reference period for two reasons. First, the month of August, after July, is the time of year when the number of aircraft movements was the highest at Montréal-Trudeau International Airport in 2019 and 2020, with respectively 22,360 and 6,336 flights (Table 1). Also, during the summer, people are more susceptible to the negative consequences of aircraft noise on their place of residence because they spend more time outside and open their windows more often. As a result, the interaction of the population with aircraft noise is more important during this time of the year rather than during winter. Due to the unavailability of data for a whole year, we use data over a short period of time, that is, one day of data collection in 2019 and another in 2020. The same procedure was used for Pretto et al. (2019) who modelled aircraft noise contours (L_{den}) of nine European airports based on data collected on a flight tracker site (FlightAware) over a period of one or two days.

For a better comparison, we chose two days with dominant winds northeast (August 14, 2019 and August 14, 2020). In this configuration, take-offs, with are generally noisier, are on runways 06L and 06R (Fig. 1) going east of the Island of Montréal (30% of the time (ADMTL, 2020a)). Aircraft taking off in this direction pass mostly over urban and residential areas, contrary to runways 24L and 24R (Fig. 1), where aircraft take off in the Lac Saint-Louis direction (70% of the time (ADMTL, 2020a)). Furthermore, in the summertime, when the air provides less aerodynamic lift, aircraft climb less quickly and fly over territories bordering the airport at low altitude. Thus, compared to a usual situation, the flight trajectories used in the modelling of aircraft noise contours can potentially impact a larger territory and become a disturbance for more individuals.

4.2.2.3. **Data required for modelling aircraft noise in IMPACT.** In the IMPACT web application, a dataset entry is minimally required to model aircraft noise in the Montréal-Trudeau International Airport vicinity. For each period studied, it is necessary to know the following information: 1) characteristics of the airport (e.g., location of the airport and weather data); 2) orientation of the runways (e.g., geographical coordinates of the runways); 3) flight trajectories (e.g., aircraft positions and flight parameters associated with each movement); 4) flight operations (e.g., number of aircraft for each runway, aircraft model, and type of operation (arrival or departure)).

Information about the characteristics of the airport and orientation of the runways were obtained on the Aéroports de Montréal (ADM) website and Google Maps, and for weather data on Environment Canada’s website. Data on flight trajectories were collected on a flight tracker website, namely FlightRadar24. The flight tracker posts real-time information on aircraft flight tracking. This Internet service collects and interprets data from thousands of ADS-B receptors located throughout the world. An aircraft equipped with an ADS-B transponder will disseminate its position but also information on its flight parameters. However, ADS-B does not provide certain data necessary for the calculation of aircraft noise, in particular flight parameters (e.g., power settings). In the present work, for each aircraft model, we have therefore used the default flight profiles (arrival or departure) from the ANP (Aircraft Noise and Performance) database offered by IMPACT. The data on flight trajectories were collected from our own ADS-B receptor built from a Raspberry Pi (Flightradar24). Then, the data were exported to CSV and KML formats from the FlightRadar24 site. Recent work has shown that data provided by flight trackers are efficient for the production of noise contour maps based on a model of best practices (Pretto et al., 2020; Pretto et al., 2019). To limit processing time, we deleted flight trajectory points above 10,000 feet. At this altitude, aircraft noise emissions are hardly audible on ground level. We also deleted incomplete or non-compliant flight trajectories. Respectively, this represents 14.49% and 8.96% of flights in 2019 and 2020. In total, 19,416 and 8,307 flight trajectory points were collected.

### Table 1

Aircraft movements\(^a\) at Montréal-Trudeau International Airport (CYUL) in August 2019 and 2020.

| Official movements\(^b\) | August 2019 | August 2020 | % Change traffic |
|---------------------------|-------------|-------------|-----------------|
| Total movements (departure, arrival) | 22,360 | 6,336 | −71.66% |
| Domestic movements | 11,816 | 4,707 | −60.16% |
| International movements | 4,155 | 980 | −78.41% |
| Transborder movements | 6,389 | 649 | −89.84% |

| Aircraft movements per runway (%)\(^c\) | August 2019 | August 2020 | Difference |
|-------------------------------|-------------|-------------|------------|
| Arrival | | | |
| 06R | 1.76% | 17.16% | +15.40 |
| 06L | 18.23% | 0.10% | −18.07 |
| 24R | 38.99% | 3.16% | −35.83 |
| 24L | 41.02% | 79.61% | +38.59 |
| Departure | | | |
| 06R | 17.27% | 17.64% | +0.37 |
| 06L | 4.45% | 0% | −4.45 |
| 24R | 47.48% | 2.77% | −44.71 |
| 24L | 30.80% | 79.50% | +48.70 |

\(^a\) According to Statistics Canada, an aircraft movement is defined as a take-off, a landing, or a simulated approach by an aircraft in the NAV Canada Air Traffic Control Manual of Operations (ATC MANOPS). \(^b\) Passenger traffic and aircraft movements (ADMTL, 2021). \(^c\) Operational indicator at YUL (ADMTL, 2020b).
respectively on August 14, 2019, and August 14, 2020 (Fig. 2).

Flight operations associated with each aircraft movement provide indication of the aircraft’s flight trajectory: vertical profile, type of aircraft, and associated runway. In total, 366 flight operations for the August 14, 2019 period and 132 for the August 14, 2020 period were identified. As a comparison, 22,360 and 6,338 aircraft movements were recorded in August 2019 and August 2020 respectively by the official data from Montréal-Trudeau International Airport (ADMTL, 2021), which represent the average daily traffic of 721 and 204. These differences with our data can partially be explained by the fact that many small planes are not equipped with an ADS-B transponder. Table 2 summarizes all flight operations of the two periods being studied.

4.3. Statistical analyses

4.3.1. Estimation of the population affected by aircraft noise

To provide a precise diagnostic of environmental equity, there should be an estimate of the populations potentially impacted by aircraft noise, that is, those residing in the various aircraft noise contour areas ($L_{den}$, $L_{dn}$, $L_{Aeq,24h}$ and $L_{Amax}$). To this end, we used the dasymetric mapping technique (Mennis, 2009). This technique consists of outlining the blocks based on noise contours by considering only the residential portion of each block (based on a land use map or satellite image, for example). We then estimated the populations impacted by multiplying the number of individuals residing in the block by the proportion of residential area of the block included in the noise zone. For example, for a block with a population of 100, where 20% of its territory is in a noise contour of $L_{den}$ 55–60 dB(A) but no residential portion in this part of the block, the total number of individuals impacted by this disturbance will be 0. This technique is implemented in the GIS software, namely QGIS 3.16 (QGIS Development Team, 2021).

4.3.2. Statistical analyses to measure environmental inequity

To answer our research question, we constructed 30 logistic regression models, respectively 15 models for the year 2019 and 15 models for the year 2020 (a regression model for each noise level - three models for each of the $L_{Aeq,24h}$, $L_{den}$, $L_{dn}$ indicators, and six models for the $L_{Amax}$ indicator). Statistical analyses were completed with version 4.0.5 of the R software (R Core Team, 2013). For these models, the observations took place with the 29,373 urban blocks, the dependent variable being whether or not they were located (1 or 0) within the noise contours of the four indicators (50–60 dB(A) with a class interval of 5 dB(A) for $L_{Aeq,24h}$, $L_{den}$, and $L_{dn}$ indicators; 65–90 dB(A) with a class interval of 5 dB(A) for $L_{Amax}$). The independent variables are the percentages of the four population groups for the blocks (low-income individuals, visible minorities, individuals 65 years and over, and children under 15 years old). Thus, the logistic regression models measure the probability that a block is or is not located in an area with a high level of aircraft noise according to the percentages of the four studied groups, for the entire Montréal census metropolitan area.

5. Results

5.1. A drastic reduction in aircraft noise footprint at Montréal-Trudeau International airport from 2019 to 2020.

Fig. 3 illustrates the noise contours of the four noise indicators in 2019 and 2020. The evolution of the aircraft noise contour areas of the different noise indicators at the Montréal-Trudeau International Airport in 2019 and 2020 is illustrated in Fig. 4. As expected, compared to 2019, the sudden drop in air traffic in 2020 caused by the global COVID-19 pandemic resulted in a drastic decrease of
noise contour areas. As an example, noise contour areas for $L_{den}^{55-60\ dB(A)}$, $L_{dn}^{55-60\ dB(A)}$, $L_{Aeq}^{24h}\ 55-60\ dB(A)$, and $L_{Amax}^{70-75\ dB(A)}$ decreased respectively by approximately 77%, 70%, 66%, and 34% during those two years. It is interesting to note, compared to cumulative noise indicators ($L_{den}$, $L_{dn}$ and $L_{Aeq}^{24h}$), the drop is smaller for the $L_{Amax}$ indicator despite higher noise levels.

Furthermore, it is noted that the territories impacted by aircraft noise are very different if we compare 2019 to 2020, in particular for the $L_{Amax}$ indicator, where we observe a change in pattern that is quite obvious in the southwest direction (Fig. 3). On the other hand, for cumulative noise indicators, there is only a decrease in the area. Overall, the $L_{Amax}$ indicator is more sensitive to the total number of movements compared to cumulative noise indicators. Indeed, $L_{Amax}$ noise contour maps were produced by combining the maximum noise level obtained for each flight event. In addition, the reduction of the aircraft noise footprint at Montréal-Trudeau International Airport, regardless of the indicator, can be explained by a differentiated use of the airport’s runways in 2019 and 2020. Indeed, in August 2020, only one runway was used (06R-24L) for take-offs and landings, unlike in 2019 where two runways were operational (06R-24L and 06L-24R). As a result, in August 2019, there were more areas overflown by aircraft compared to 2020, especially east of the airport (i.e., above the city rather than above Lac Saint-Louis).

### Table 2

Information about the noise model used to generate noise contours at Montréal-Trudeau International Airport (CYUL) in 2019 and 2020.

| Parameter | Model movements | | % Change traffic |
|-----------|-----------------|-----------------|------------------|
| Total movements download$^a$ (departure, arrival) | 428 | 145 | −66.1% |
| Movements in the model (departure, arrival) | 366 | 132 | −63.9% |
| Day movements (6 a.m. to 6 p.m.) | 228 | 83 | −63.6% |
| Evening movements (6 p.m. to 10 p.m.) | 109 | 42 | −61.5% |
| Night movements (10 p.m. to 6 a.m.) | 30 | 6 | −80.0% |
| Operational runways | 4 | 1 | −50.0% |
| Prevailing winds$^a$ | N-NE | N-NE | |

| Aircraft weight class$^b$ | | | |
|---------------------------|-----------------|-----------------|
| August 14, 2019 (%) | August 14, 2020 (%) | Difference |
| Small | 0% | 2% | +2 |
| Large | 73% | 73% | 0 |
| Heavy | 27% | 26% | −1 |

| Aircraft noise chapter$^c$ | | | |
|---------------------------|-----------------|-----------------|
| August 14, 2019 (%) | August 14, 2020 (%) | Difference |
| Chapter 2 (<1977) | 0% | 2% | +2 |
| Chapter 3 (1977-2006) | 97% | 86% | −11 |
| Chapter 4 (2006-2017) | 3% | 13% | +10 |

| Stage length (distance to fly in Nm)$^d$ | | | |
|-------------------------------|-----------------|-----------------|
| August 14, 2019 (%) | August 14, 2020 (%) | Difference |
| >3500 Nm | 8.5% | 6.1% | −2.4 |

| Aircraft movements per runway (%) | | | |
|----------------------------------|-----------------|-----------------|
| August 14, 2019 (%) | August 14, 2020 (%) | Difference |
| Arrival | | | |
| 06R | 0.64% | 46.21% | +45.57 |
| 06L | 53.50% | 0 | −53.50 |
| 24R | 19.75% | 0 | −19.75 |
| 24L | 26.11% | 0 | −26.11 |
| Departure | | | |
| 06R | 59.33% | 53.79% | −5.54 |
| 06L | 11.96% | 0 | −11.96 |
| 24R | 28.23% | 0 | −28.23 |
| 24L | 0.48% | 0 | −0.48 |

$^a$: Data downloaded in flight tracking websites (Flightradar24). $^b$: Wake turbulence separation minima is based on a grouping of aircraft types into three categories according to maximum certified take-off mass (MTOM) as follows: Small (aircraft have a certificated takeoff weight less than 41,000 lb.); Large (aircraft have a certificated takeoff weight between 41,000 lb. (18,600 kg) and 255,500 lb.); Heavy: (any aircraft with takeoff weight more than 255,500 lb. (116,000 kg)). This classification is by the FAA. $^c$: Aircraft noise certification according to ICAO regulations, Annex 16, Volume 1. Particularly complex, the higher the chapter on noise, the more stringent the noise standards for aircraft (ICAO, 2008). $^d$: Trip range in nautical miles (distance to fly). The stage length value is further used by IMPACT to select an associated aircraft default weight from the ANP database.
Fig. 3. Noise contours in 2019 and 2020 for the four noise indicators.
Fig. 4. Areas of noise contours in 2019 and 2020 according to noise indicator.

Fig. 5. Total population according to the noise level of the four indicators in 2019 and 2020.
5.2. A decrease in populations exposed to aircraft noise from 2019 to 2020

The noise contour areas of the four noise indicators were weaker in 2020; it is therefore not surprising to find a narrower range of population numbers compared to 2019. The results show a decrease of 78.3%, 82.8%, 84.9%, and 17.9% respectively for noise contours $L_{den}$ 55–60 dB(A), $L_{den}$ 55–60 dB(A), $L_{den}$ 55–60 dB(A), and $L_{den}$ 70–75 dB(A). Compared to the three cumulative indicators for aircraft noise exposure ($L_{den}$, $L_{dn}$ and $L_{dn}$), the high numbers observed for the single-event indicator ($L_{den}$) can be explained by much greater noise contours in 2019 and 2020 (respectively 440 km$^2$ and 266 km$^2$ for the $L_{den}$ 65–70 dB(A) zone, compared to 10 km$^2$ and 3 km$^2$ for the $L_{den}$ ≥ 65 dB(A) zone). The estimates are included in Fig. 5 and in the supplementary material (Tables A1 and A2).

5.3. Effect of the reduction in size of the noise contours at Montréal-Trudeau International airport on environmental equity

The results of the 30 logistic regression models allow us to provide an environmental equity diagnostic for the four groups presented in Table 3. Due to lack of space, only the odds ratios (OR) values are reported. Note that the complete output of these models is reported in the supplementary material (Table A3). The odds ratios can be interpreted as follows: an odds ratio greater than 1 indicates a situation of inequity, while an odds ratio less than 1 indicates an advantageous situation. The further the odds ratios are from 1, the greater the effect; values in bold type are significant at $p = 0.05$ (Table 3). The results show that environmental equity patterns vary according to type of indicator and noise levels used, but also according to year. That said, the situation of visible minorities is quite different from the other three groups since almost all odds ratios values are significantly greater than 1.

| Table 3 | Odds ratios for the logistic regression according to noise indicators. |
|---------|---------------------------------------------------------------------------------|
| $L_{den}$ (2019) | $L_{den}$ (2020) |
| 50–55dB | 55–60dB | ≥60dB(A) | 50–55dB | 55–60dB | ≥60dB(A) |
| 0–14 years old (%) | 0.981 | 1.049 | 0.961 | 1.043 | 0.991 | 1.040 |
| 65 and over (%) | 0.997 | 1.044 | 1.022 | 1.042 | 1.022 | 1.041 |
| Low-income pop. (%) | 1.016 | 0.967 | 0.932 | 0.976 | 0.953 | 0.929 |
| Visible minorities (%) | 1.038 | 1.060 | 1.059 | 1.061 | 1.055 | 1.015 |
| $L_{den}$ (2019) | $L_{den}$ (2020) |
| 50–55dB | 55–60dB | ≥60dB(A) | 50–55dB | 55–60dB | ≥60dB(A) |
| 0–14 years old (%) | 0.968 | 1.031 | 0.993 | 1.020 | 1.041 | 0.992 |
| 65 and over (%) | 0.996 | 1.041 | 1.029 | 1.037 | 1.043 | 1.023 |
| Low-income pop. (%) | 1.007 | 0.954 | 0.915 | 0.964 | 0.923 | 0.938 |
| Visible minorities (%) | 1.042 | 1.064 | 1.059 | 1.062 | 1.048 | 1.012 |
| $L_{dn}$ (2019) | $L_{dn}$ (2020) |
| 50–55dB | 55–60dB | ≥60dB(A) | 50–55dB | 55–60dB | ≥60dB(A) |
| 0–14 years old (%) | 1.036 | 0.957 | 0.847 | 0.993 | 1.057 | 1.122 |
| 65 and over (%) | 1.036 | 1.018 | 0.993 | 1.023 | 1.050 | 1.072 |
| Low-income pop. (%) | 0.972 | 0.929 | 0.891 | 0.957 | 0.915 | 0.957 |
| Visible minorities (%) | 1.059 | 1.059 | 1.049 | 1.057 | 1.017 | 1.036 |
| $L_{dn}$ (2019) | $L_{dn}$ (2020) |
| 65–70dB | 70–75dB | 75–80dB | 80–85dB | 85–90dB | ≥90dB(A) |
| 0–14 years old (%) | 0.926 | 0.987 | 0.985 | 0.988 | 0.942 | 0.822 |
| 65 and over (%) | 0.991 | 1.001 | 1.006 | 1.021 | 1.015 | 0.982 |
| Low-income pop. (%) | 1.065 | 1.037 | 0.990 | 0.942 | 0.900 | 0.874 |
| Visible minorities (%) | 1.032 | 1.033 | 1.049 | 1.056 | 1.044 | 1.050 |
| $L_{dn}$ (2020) | $L_{dn}$ (2020) |
| 65–70dB | 70–75dB | 75–80dB | 80–85dB | 85–90dB | ≥90dB(A) |
| 0–14 years old (%) | 0.993 | 1.024 | 1.038 | 0.990 | 1.068 | 1.098 |
| 65 and over (%) | 1.021 | 1.019 | 1.040 | 1.022 | 1.049 | 1.061 |
| Low-income pop. (%) | 1.065 | 1.029 | 0.985 | 0.950 | 0.900 | 0.976 |
| Visible minorities (%) | 1.055 | 1.045 | 1.058 | 1.055 | 1.024 | 1.022 |

Values in bold are significant at the level of $p = 0.05$. See the supplementary material (Table A3) for the complete output of the models including the values of 95% confidence interval and fit statistics (AIC, and pseudo r-squared).
In 2019, visible minorities are in a situation of inequity in all noise contours: \( L_{den} \) (OR = 1.038, 1.060, 1.059 for 50–55, 55–60 and ≥ 60 dB(A) models respectively), \( L_n \) (OR = 1.042, 1.064 and 1.059), \( L_{Aeq,24h} \) (OR = 1.059, 1.059 and 1.049), and \( L_{Amax} \) (OR = 1.032, 1.033, 1.049, 1.056 and 1.050 for 65–70, 70–75, 75–80, 80–85, 85–90 and ≥ 90 dB(A) respectively). To a lesser extent, this finding also applies to seniors for several noise contours: \( L_{den} \) (OR = 1.044 and 1.022 for 55–60 and ≥ 60 dB(A) respectively), \( L_n \) (OR = 1.041 and 1.029 for 55–60 and ≥ 60 dB(A) respectively) and \( L_{Aeq,24h} \) (OR = 1.036 and 1.018 for 50–55 and 55–60 dB(A) respectively), as well as \( L_{Amax} \) (OR = 1.021 for 80–85 dB(A)).

The situation is more complex for two other groups, both favourable and unfavourable depending on indicator and noise level used. Indeed, there are favourable situations for children in several noise contours of the four indicators: \( L_{den} \) and \( L_n \) (OR = 0.981 and 0.968 for 50–55 dB(A)), \( L_{Aeq,24h} \) (OR = 0.957 and 0.847 for 55–60 and ≥ 60 dB(A) respectively), and \( L_{Amax} \) (OR = 0.987, 0.942 and 0.882 for 70–75, 85–90 and ≥ 90 dB(A) respectively). Low-income populations are also in an advantageous situation in some noise contours: \( L_{den} \) (OR = 0.967 and 0.932 for 55–60 and ≥ 60 dB(A) respectively), \( L_n \) (OR = 0.954 and 0.915 for 55–60 and ≥ 60 dB(A) respectively), \( L_{Aeq,24h} \) (OR = 0.972, 0.929 and 0.891 for 50–55, 55–60 and ≥ 60 dB(A) respectively), and \( L_{Amax} \) (OR = 0.990, 0.942, 0.900 and 0.874 for 75–80, 80–85, 85–90 and ≥ 90 dB(A) respectively).

Conversely, children are in a situation of inequity at a noise level of 55–60 dB(A) for the \( L_{den} \) and \( L_n \) indicators (OR = 1.049 et 1.031), and at a level of 50–55 dB(A) for the \( L_{Aeq,24h} \) (OR = 1.036). Similarly, low-income populations are in a situation of inequity, although not very pronounced, for both \( L_{den} \) and \( L_n \) noise contours at the 50–55 dB(A) (OR = 1.016 and 1.007 levels, but also for \( L_{Amax} \) noise contours at the 65–70 and 70–75 dB(A) (OR = 1.065 and 1.037) levels.

In 2020, we see some significant changes to the equity patterns compared to 2019, especially for children and seniors. First, notwithstanding the noise level, children are not more significantly advantaged for the three cumulative noise indicators: \( L_{den} \), \( L_n \) and \( L_{Aeq,24h} \), and the single-event indicator \( \left( L_{Amax} \right) \). The situation even worsens at certain levels of noise for the \( L_{den} \) (OR = 1.043 for 50–55 dB(A) model) and \( L_{Amax} \) indicators (OR = 1.024 and 1.038 for 70–75 and 75–80 dB(A) models respectively). As for seniors, we see that solely for the \( L_{Amax} \) indicator, does the situation worsen for this group and almost at all levels (OR = 1.021, 1.019, 1.040, 1.022 and 1.049 for 65–70, 70–75, 75–80, 80–85 and 85–90 dB(A) models respectively). Finally, the situations of inequity for visible minorities and low-income populations do not vary much between the two years, exception of a few non-significant values in 2020.

6. Discussion

6.1. Limitations of the study

Many limitations inherent to this research must be mentioned. First, this study is based on concentration measures which are not exposure measures. Indeed, we are only interested in the concentration of aircraft noise found in residential areas in the Montréal CMA, where individuals spend many hours of their day, and this is especially true during the COVID-19 pandemic where telework is mandatory. In other words, we are not informed on actual exposure for individuals during a specific period of time. It would therefore be interesting in future studies to use individual exposure measures based on noise measuring devices in real time. Furthermore, for a more precise estimate of the populations affected by aircraft noise, it would be interesting in future studies to take into account individual human mobility patterns. Indeed, by considering only the residential areas, we illustrate only a part of the populations potentially affected by aircraft noise.

As for noise indicators used, noise predictions were made during the two busiest days in the summers of 2019 and 2020, with take-offs mainly heading east of the Island of Montréal in densely populated sectors. These two days are therefore supposed to be representative of the worst days of the year in terms of noise emissions. Nonetheless, for a better comparison, it would have been interesting to take other scenarios into account, especially that of take-offs heading west of the island toward Lac Saint-Louis. The noise impact would have surely been different. Moreover, in this study, we used default flight profiles to model aircraft noise. However, these data are generic and do not realistically represent certain flight operations at Montréal-Trudeau International Airport. Finally, considering the difference between the number of flight events collected and the official data on the average daily aircraft movements, the aircraft noise levels in 2019 and 2020 are probably underestimated in this study.

6.2. The impacts of the type of aircraft noise indicator used on environmental equity assessment.

In light of the results, the estimates of populations impacted by levels of aircraft noise may differ from one indicator to the next. Our results corroborate those of the Montréal public health advisory (Smargiassi et al., 2014) showing that the choice of aircraft noise indicators can lead to different estimates. As for the results of the regression models, we observe two types of situations in terms of environmental equity according to the type of indicator used.

For cumulative noise indicators \( (L_{den}, L_n, L_{Aeq,24h}) \), the environmental equity patterns vary slightly according to type of indicator and noise contours, but also from one year to the next, with the exception of visible minorities where there is little change between 2019 and 2020. This corroborates, in part, the findings advanced by Ogneva-Himmelberger and Cooperman (2010) in Boston. It is also found that the areas affected by high levels of aircraft noise are not necessarily the disadvantaged sectors, in 2019 as well as in 2020. These results corroborate those reported by Audrin et al. (2019) in the Montréal metropolitan area. Conversely, for visible minorities and to a lesser extent for seniors, the situation is more inequitable in 2019 and in 2020 for most of the noise contours used. For children, the situation is more complex for certain indicators and if both years are compared.

For the single-events noise indicator \( L_{Amax} \), differentiated situations are found for children and low-income individuals. It is particularly noted that between 2019 and 2020, the situation is inequitable for low-income individuals (at noise levels of 65–70 and
70–75 dB(A) $L_{Amax}$), whereas the situation deteriorates for children from one year to the next.

The geographical dimension could be part of the answer. For cumulative noise indicators, the sectors exposed to high noise levels (55–60 and ≥ 60 dB(A)) are mostly located in proximity to urban areas in municipalities adjacent to the Montréal-Trudeau International Airport (Dorval and Pointe-Claire to the west, and the Saint-Laurent district (Bois-Franc neighbourhood) and Ville Mont-Royal to the east). These sectors accommodate mostly wealthy households, owners of single-family multi-storey homes with garages (cottage or bungalow types) and whose purchasing cost is high (Audrin et al., 2019). However, for the high noise levels for the $L_{Amax}$ indicator (65–70 and 70–75 dB(A)), but also at lower noise levels for the $L_{den}$ and $L_{dn}$ indicators (50–55 dB(A)), noise contour areas being greater, within these zones there are certain central neighbourhoods of the Island of Montréal (Villeray–Saint-Michel–Parc-Extension, Côte-des-Neige-Notre-Dame-de-Grâce), as well as surrounding neighbourhoods (Saint-Léonard, Lachine and Ahuntsic-Cartierville), where we find, at least in some parts of these territories, a high proportion of the low-income population (Adès et al., 2012; Séguin et al., 2012). This can contribute to the presence of inequity for low-income individuals. However, let us mention that, at very high levels of noise ($L_{Amax}$ ≥ 75 dB(A)), the situation improves for this group.

As for members of visible minorities, the inequity measured for this population group in 2019 and in 2020, notwithstanding the indicator and noise levels, can be explained by their high numbers in certain surrounding neighbourhoods and in the close suburbs of the Montréal metropolitan area. The presence of visible minorities is significant in certain neighbourhoods of the City of Montréal, and more particularly in sectors located in the vicinity of the airport (Saint-Laurent district) and sectors further away but partly located within the noise contour locations (the districts of Villeray–Saint-Michel–Parc-Extension, Ahuntsic-Cartierville, and Ville Mont-Royal). These residential areas are characterized by both a strong proportion of visible minorities, but also recent immigrants (Apparicio et al., 2008; Leloup, 2007).

For children, they are in a situation of inequity for high noise levels of $L_{den}$ and $L_{dn}$ (55–60 dB(A)), but not at lower levels (50–55 dB(A)). The size of the noise contours being smaller at high levels of aircraft noise, we mainly find sectors of the inner suburbs within these zones. The high proportion of families with children outside the central neighbourhoods (Apparicio et al., 2010) explains why this population group is overrepresented in areas exposed to high levels of road traffic.

For seniors, they are in a situation of inequity for many noise indicators in 2019 and 2020. These results are surprising given that few studies in environmental equity have considered this population group. This situation can be explained by the overrepresentation of this group outside Montréal’s central neighbourhoods (Séguin et al., 2013), in sectors characterized by a strong presence of private seniors residences (Carrier et al., 2013). Considering these results, this group should be more often included in the work on environmental equity related to aircraft noise.

7. Conclusion

In this article, we have presented an environmental equity diagnostic within a longitudinal perspective for four population groups as to the level of aircraft noise measured in their residential area, using four noise indicators ($L_{den}$, $L_{dn}$ $L_{Aeq,24h}$, and $L_{Amax}$). We must remember that, in Canada, travel restrictions (e.g., border shutdowns) implemented by the federal government to prevent the spread of COVID-19 in 2020 had an effect on air traffic, which drastically decreased aircraft noise footprints, namely, the scope of noise contours in the vicinity of the Montréal-Trudeau International Airport. It should be noted that the change in patterns of aircraft noise contours in 2020 compared to 2019 is related not only to the decrease in air traffic, but also to the restrictions of flight movements on certain runways of the Montréal-Trudeau International Airport. However, the results observed in Montréal show that, despite a significant decrease in population, notwithstanding the group impacted by aircraft noise in 2020 compared to 2019 (e.g., at noise levels of 55–60 dB(A) $L_{den}$, 94,651 and 20,533 individuals are exposed to noise in 2019 and 2020 respectively, that is, a decrease of 78.3%), this does not necessarily lead to a decrease in inequities for seniors and members of visible minorities.

Furthermore, this research revealed that using a specific indicator (cumulative or single-events) can lead to different results for the same situation in terms of estimates of impacted populations and environmental equity since the two types of noise indicators are not comparable. The single-events indicator ($L_{Amax}$) only measures the maximum level of noise; it is therefore very sensitive to each flight event (Pretto et al., 2020). This indicator thus reflects the short-term disturbance generated by one-time noise events such as a large-passerenger aircraft. Conversely, the cumulative noise indicators ($L_{den}$, $L_{dn}$ and $L_{Aeq,24h}$) represent weighted averages over a period of 24 h and are rather well adapted to deal with disturbances experienced on the long term, especially for sources of continuous noise such as road traffic.
One of the contributions of this article to current literature in environmental equity is that the choice of aircraft noise indicators can be a determining factor since it can potentially influence results. In future studies, using different indicators would more accurately reflect the noise impact of aircraft traffic. In fact, in most countries, regulation pertaining to aircraft sound disturbances are mostly based on cumulative indicators (average noise over a specific time period) to characterize noise. As a complement to cumulative indicators, the use of single-events indicators that address maximum sound levels must be favoured in future studies on environmental equity, but also in diagnostic and monitoring studies. The latter are considered to be more representative of the disturbance experienced by the surrounding population. Furthermore, this work has highlighted that, according to trajectories and number of aircraft movements in a given airport, this can have an influence on noise footprints (or noise contours) on the ground. It is therefore important to regularly update noise contour maps to precisely evaluate the impacts of aircraft noise disturbances, particularly when certain events can influence the use of runways, number of movements, aircraft fleet configuration, or again, flight profiles at a given airport (e.g., COVID-19, works on airport runways, specific weather conditions, etc.). To this effect, for future work on environmental equity, it is important to take into consideration the longitudinal effects of aircraft noise. However, in Canada, there is no directive for the mandatory establishment of updated mapping on a regular basis, contrary to Europe. The current measures in terms of ground use and management in the vicinity of Canadian airports seem to be limited. In the future, aviation decision-makers will have to bring particular attention to the post-health crisis because the recovery of air traffic could exacerbate the disturbances experienced by populations residing in the vicinity of airports since they have become used to living in less noisy environments.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.trd.2022.103274.

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