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The healthcare costs of poor air quality in Ireland: An analysis of hospital admissions

Anne Nolan[a,b*]

a) Economic and Social Research Institute, Dublin, Ireland
b) Department of Economics, Trinity College Dublin, Dublin, Ireland

*Corresponding Author:
Dr Anne Nolan
Economic and Social Research Institute,
Whitaker Square, Sir John Rogerson's Quay,
Dublin, Ireland
Email: Anne.Nolan@esri.ie

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The Healthcare Costs of Poor Air Quality in Ireland: An Analysis of Hospital Admissions[1]

Anne Nolan[1,2]

[1] Economic and Social Research Institute, Dublin

[2] Trinity College, Dublin

Abstract

Air pollution is well recognised as a major risk factor for disease and premature mortality worldwide. In Ireland, fine particulate matter ($PM_{2.5}$), which originates largely from burning solid fuel for heating, and nitrogen dioxide (NO_2), derived from road transport, are the main sources contributing to poor air quality. Although average annual air pollution concentrations have decreased in Ireland over the past two decades, there are concerns about exceedances above WHO air quality guidelines (AQGs) in many cities and towns in Ireland. In this paper, we estimate the acute healthcare costs of air pollution in Ireland, using data on emergency inpatient hospital admissions and costs over the period 2016-2019 from the Hospital In-Patient Enquiry (HIPE) system, supplemented with data on population attributable fractions for specific conditions with causal links to air pollution (e.g., asthma). Quantifying the potential impact of air pollution on healthcare costs is important for future policy and resource planning and for targeting of mitigation measures and public health campaigns. Over the four-year period we examine, the acute healthcare costs of treating five conditions attributable to ambient air pollution (asthma in children, and chronic obstructive pulmonary disease, ischaemic heart disease, stroke and asthma in adults) amounted to €56.0m (range €15.0m to €105.8m). Hospitalisations attributable to air pollution (mainly $PM_{2.5}$) for these five conditions accounted for 63,572 bed days (range 17,767 to 115,996) over the period 2016-2019. In terms of the policy response, the recent Clean Air Strategy commits to achieving the final WHO air quality guideline (ACQ) values by 2040. Achieving these targets will require continued policy focus on measures such as moving away from the burning of solid fuels. In addition, policy measures to mitigate the impacts of climate change, such as decarbonising home heating, promoting active travel and transitioning to electric vehicles, will have concomitant benefits for air quality and population health.

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1. Background

Air pollution is well recognised as a major risk factor for disease and premature mortality worldwide (European Environment Agency 2022; Murray et al. 2020; Vos et al. 2020). Global assessments of ambient (outdoor) air pollution suggest that between 4 million and 9 million deaths annually and hundreds of millions of lost years of healthy life can be attributed to ambient air pollution (WHO 2021). The global burden of disease attributable to air pollution is now estimated to be comparable with other major health risks such as unhealthy diet and tobacco smoking, and was in the top five out of 87 risk factors for male and female deaths in 2019 (Murray et al. 2020).[2] As a result, air pollution is now recognised as the single largest environmental threat to public health (WHO 2021).

Ambient air pollution is a complex mixture of particles and gases. Their concentrations and composition vary from place to place, depending on what sources are present, weather conditions, and how they mix in the atmosphere (Burns et al. 2020). The European Union (EU) Ambient Air Quality Directives (currently under review) set EU air quality standards for 12 air pollutants: sulphur dioxide, nitrogen dioxide/nitrogen oxides, particulate matter (PM_{10}, $PM_{2.5}$), ozone, benzene, lead, carbon monoxide, arsenic, cadmium, nickel, and benzo(a)pyrene. In 2021, the World Health Organization (WHO) published new air quality guidelines (AQGs) for particulate matter ($PM_{2.5}$ and PM_{10}), ozone, nitrogen dioxide, sulfur dioxide and carbon monoxide following a systematic review of the latest scientific evidence demonstrating how air pollution damages human health (WHO 2021).[3]

In Ireland, fine particulate matter ($PM_{2.5}$), which originates largely from burning solid fuel for heating, and nitrogen dioxide (NO_2), derived from road transport, in particular from diesel engines, are the main sources contributing to poor air quality (EPA 2022). Although air pollution has decreased in most

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[2] The global burden of disease (GBD) methodology has a risk factor hierarchy. Level 1 risk factors are behavioural, environmental and occupational, and metabolic; Level 2 risk factors include 20 risks or clusters of risks (e.g., air pollution); Level 3 includes 52 risk factors or clusters of risks; and Level 4 includes 69 specific risk factors (e.g., ambient particulate matter). Counting all specific risk factors and aggregates computed in GBD 2019 yields 87 risks or clusters of risks. In the GBD methodology, air pollution is a level 2 risk, comprised of particulate matter air pollution (level 3) and ozone pollution (level 3). Particulate matter pollution is further comprised of ambient particulate matter pollution (level 4) and household particulate matter pollution (level 4) (Murray et al. 2020). However, it is important to note that, as of 2021, global disease burden estimates are limited to $PM_{2.5}$ and ozone. Other common pollutants such as nitrogen dioxide and sulphur dioxide are not yet included and, therefore, these figures based on exposure to $PM_{2.5}$ and ozone are likely to underestimate the full health toll from air pollution (WHO 2021).

[3] For example, the WHO updated guidelines for $PM_{2.5}$ are:

- annual average of 5 $\mu g/m^3$ (previous limit was 10)
- daily average of 15 $\mu g/m^3$ (previous limit was 25)

European countries over the past two decades, including Ireland, levels of ambient air pollution remain above WHO AQGs in many cities and towns in Ireland (EPA 2022). For example, in 2021, 61/81 $PM_{2.5}$ monitoring stations exceeded the new WHO daily AQG for $PM_{2.5}$ and 65/81 stations exceeded the new WHO annual AQG (EPA 2022). In Ireland, the current Clean Air Strategy commits to achieving the interim WHO AQG IT3[4] target by 2026, the IT4 target by 2030 and the achievement of final WHO AQG values by 2040 (Government of Ireland 2023).

As noted, air pollution is now well recognised as a major risk factor for disease and premature mortality worldwide. The greatest health damage from ambient air pollution is caused by chronic exposure to particulate matter, in particular to $PM_{2.5}$ which increases the risk of heart diseases, stroke, lung cancer and many respiratory diseases including asthma, bronchitis, chronic obstructive pulmonary disease (COPD) and respiratory infections (Brook et al. 2010; Cohen et al. 2017; OECD 2016, 2020). Air pollution exposure may increase the incidence of, and mortality from, a larger number of diseases and conditions than those currently considered, such as cognitive impairment and dementia (Ailshire and Crimmins 2014; Peters et al. 2019; Weuve et al. 2012; Wood et al. 2022), lung cancer (Pimpin et al. 2018), type 2 diabetes and neonatal mortality (Murray et al. 2020). There is also a growing evidence base linking air pollution (particularly $PM_{2.5}$) with poorer mental health and wellbeing, including depression and anxiety (Braithwaite et al. 2019; Power et al. 2015), suicide (Gładka et al. 2021), bipolar disorder (Hao et al. 2022) and life satisfaction (Orru et al. 2016).[5] Air pollution may also exacerbate existing conditions; for example, it can worsen the prognosis of pneumonia patients (Yee et al. 2021). Certain population subgroups are particularly vulnerable to the effects of air pollution, such as children and older people (Neidell 2004; Nhung et al. 2017). In addition, for a given level of air pollution, those in more disadvantaged socioeconomic positions may be more susceptible to the negative health consequences of air pollution (due to pre-existing health conditions, poorer housing conditions, etc.) (European Environment Agency 2018; OECD 2020).

In addition to the mortality and morbidity burden, air pollution also imposes a significant economic burden, in terms of healthcare costs, lost productivity, impact on agricultural crops and damage to buildings and infrastructure (OECD 2020; WHO 2021). The OECD estimate that the total welfare losses

[4] There are four Interim Targets (IT) identified (IT1, IT2, IT3, IT4) for each pollutant. For example, for annual $PM_{2.5}$, IT3 is 15 $\mu g/m^3$ and IT4 is 10 $\mu g/m^3$.

[5] (Aguilar-Gomez et al. 2022) survey the growing literature within economics that has begun to investigate the causal effects of air pollution on numerous 'non-health' outcomes, such as worker productivity, school performance, decision-making, and even crime.

from ambient air pollution ($PM_{2.5}$ and ground-level ozone) in the EU-27 in 2017 amounted to €601bn, or approximately 4.9 per cent of GDP. While the majority (88 per cent) of the welfare losses were accounted for by premature mortality, healthcare costs[6] amounted to €15bn (or 2.5 per cent of the total welfare losses). In Ireland, the healthcare costs were estimated at €0.15bn, or 0.06 per cent of GDP[7] (OECD 2016, 2020). In England, it was estimated that the total health and social care costs of $PM_{2.5}$ in 2017 were £41.2m (based on data where there is more robust evidence for an association), increasing to £76.1 million when diseases are included where the evidence is associative or emerging (Pimpin et al. 2018; Public Health England 2018).[8]

The existing literature uses a variety of methods to examine the link between air pollution and healthcare utilisation in order to inform analyses of the healthcare costs of air pollution. Using individual-level data, a large body of evidence has assessed the association between air pollution and healthcare (usually hospital) utilisation. A common approach is the use of daily time-series data on outcomes, such as hospitalizations, linked with contemporaneous and lagged levels of pollution and potential confounding variables, such as weather (Anderson et al. 2003; Carugno et al. 2016; Chen et al. 2022; Dominici et al. 2006; Liu et al. 2018; Xu et al. 2016; Zanobetti and Schwartz 2006; Zhou et al. 2023).[9] In Ireland, (Clancy et al. 2002) and (Dockery et al. 2013) used this approach to examine the impact of the so-called 'smoky coal bans' on mortality and hospitalisations during the 1990s.[10] However, analyses of the hospital admissions data at that time were hampered by substantial

[6] Healthcare costs relate to the cost of treating lung cancer, cardiovascular and respiratory diseases due to high concentrations of $PM_{2.5}$ and ozone (OECD 2016).
[7] Caution must be exercised in making international comparisons of data using GDP as the denominator for Ireland, as GDP figures for Ireland are heavily influenced by the activities of global multinational companies locating in Ireland (Wren and Fitzpatrick 2020).
[8] They examined primary care visits, prescriptions, secondary care (inpatient and outpatient) visits, and social care. Asthma, COPD, coronary heart disease, stroke, type 2 diabetes, and lung cancer were included within the model based on estimates of associations between exposure to the pollutants and risk of developing the diseases, which were obtained from meta-analyses of prospective cohort studies. Low birth weight and dementia were also included, although the evidence was less well established for these conditions.
[9] See (Adar et al. 2014), (Yee et al. 2021), (Zheng et al. 2015) and (Zhu et al. 2020) for meta-analyses, and (Brook et al. 2010) for an overview of studies examining cardiovascular mortality and hospitalisations.
[10] The two studies found divergent results for cardiovascular mortality. (Dockery et al. 2013) found that respiratory mortality decreased significantly, by 17 per cent, after the 1990 ban (confirming the earlier study by Clancy et al., 2002) and, to a lesser extent, after the 1995 and 1998 bans. However, unlike the earlier study, they did not find a reduction in total or cardiovascular mortality after either the 1990 ban or the later bans. The authors concluded 'we now believe the previous analyses (Clancy et al. 2002) overestimated the Dublin ban's effects on mortality rates for those causes with substantial long-term trends, that is, total and cardiovascular mortality'.

underreporting issues and the absence of data from a reference population to account for long-term background trends.[11]

An alternative approach is the use of cohort data that follows individuals over time and compares pollution measures aggregated over time with health outcomes (and healthcare utilisation) (Gan et al. 2013; Wood et al. 2022). Both the time-series and cohort approaches have been criticised as ambient air pollution is not randomly assigned across the population, leading to an increasing interest in quasi-experimental techniques to isolate exogenous changes in pollution (see (Alexander and Schwandt 2022; Brook et al. 2010; Deryugina et al. 2019; Lleras-Muney 2010; Moretti and Neidell 2011; Neidell 2004, 2009; Ward 2015) for good examples).[12] A recent quasi-experimental analysis of the extension of smoky coal bans to smaller towns in Ireland over the period 2010-2018 found significant negative effects of the bans on the incidence of chronic lung disease among the older population, but non-significant effects on all-cause mortality (Lyons et al. 2023).

In this paper, we build on this literature to estimate the acute healthcare costs of air pollution in Ireland, using data on hospital admissions and costs over the period 2016-2019. Quantifying the potential impact of air pollution on healthcare costs is important for future policy and resource planning and for targeting of mitigation measures and public health campaigns (Pimpin et al. 2018). We use an approach that does not rely on the availability of the data summarised above, such as linked high-frequency data on air pollution concentrations and healthcare utilisation. In Ireland, while the number of air quality monitors is now at 116 (much increased from just a few years ago) (EPA 2022), the challenge is being able to link these data to appropriate high-frequency spatially-coded administrative data on healthcare utilisation and costs. Low monitor coverage may also mean that there may be significant variation in local air pollution concentrations within the area covered by a single monitoring station. In addition, the time series approach can only identify the short-term effects of air pollution, while cohort-based approaches largely focus on the longer-term impacts. Previous research has also highlighted the difficulty in attributing ambient air pollution data to individuals; in

[11] Regular reporting of hospital admissions began in 1990; no data were available before the 1990 coal ban in Dublin, and only limited amounts of data were available before the 1995 and 1998 bans.

[12] (Aguilar-Gomez et al. 2022) also discuss the trade-offs involved in using data on air pollution in empirical research; they note that in general, the finer the temporal scale, the coarser is the spatial scale (and vice versa). For example, reliable daily global measures of air pollution are available at a 50x62.5km grid, and global annual surface $PM_{2.5}$ available at resolutions as fine as 1kmx1km grid (van Donkelaar et al. 2016; Hammer et al. 2022).

particular, personal exposure will also depend on indoor pollution as well as on individual behaviour (mobility and time spent outdoors) (Lleras-Muney 2010).

Our approach, using data on hospital admissions and costs, supplemented with data on the burden of ill-health attributed to air pollution for specific conditions with causal links to air pollution (e.g., asthma), circumvents these types of issues.[13] The approach is similar to that used in cost-of-illness studies, in which the costs to society of a particular disease are quantified. However, while a full cost-of-illness study would include the direct costs of treating the disease such as healthcare costs for diagnosis, treatment and management of disease progression and patients' own costs (travel, over-the-counter medication)[14], as well as indirect costs such as productivity loss resulting from time off employment, we focus here on the acute hospital care costs only. Section 2 describes the data and methodology in greater detail, Section 3 presents the empirical results, while Section 4 discusses the results and implications for policy.

2. Data and Methods

a. Hospital In-Patient Enquiry (HIPE)

The main source of data used in this paper is sourced from the Hospital In-Patient Enquiry (HIPE) database, administered by the Healthcare Pricing Office (HPO). The HIPE is a health information system designed to collect clinical and administrative data on discharges from, and deaths in, acute public hospitals in Ireland. The data cover day and inpatient (elective, emergency and maternity) discharges for all public hospitals in Ireland (n=53 in 2019).[15] A HIPE discharge record is created when a patient is discharged from (or dies in) hospital. This record contains administrative, demographic and clinical information for a discrete episode of care. An episode of care begins at admission to hospital, as a day or inpatient, and ends at discharge from (or death in) that hospital. Due to the absence of a unique patient identifier in the Irish healthcare system, it is not possible to follow activity at the patient level (that is, attribute multiple discharges to the same patient) across hospitals (Keegan et al. 2020).

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[13] A similar approach has been used to estimate the acute hospital costs of water-related diseases in Ireland (Griffin and Walsh 2022).
[14] https://yhec.co.uk/glossary/cost-of-illness/
[15] Private hospital activity is not captured in HIPE. Most (over 90 per cent) of private hospital activity in Ireland is financed by private health insurance. In 2018, it was estimated that the three main private health insurers together paid almost 500,000 day patient claims and 110,000 in-patient claims (Keegan et al. 2022). In 2018, the HIPE system reported 1.1m day patients, and 0.651m in-patients (Healthcare Pricing Office 2019).

For each discharge, a principal diagnosis and up to 29 secondary diagnoses are recorded. At the start of 2020, the classification used to code clinical information in Ireland was updated from the 8th edition (in use since 1st January 2015) to the 10th edition of the International Statistical Classification of Diseases and Related Health Problems, Australian Modification (ICD-10-AM), Australian Classification of Health Interventions (ACHI) and Australian Coding Standards (ACS).[16] ICD-10-AM codes are organised into 22 chapters, which reflect the main underlying disease category. For example, respiratory diseases are coded to chapter 'J', with further disaggregation to a three character code to identify specific diagnoses (e.g., asthma is identified using the code 'J45').

In order to analyse the acute healthcare costs of air pollution-related hospital discharges, the Diagnosis Related Group (DRG) scheme enables the disaggregation of discharges into homogenous groups, which undergo similar treatment processes and incur similar levels of resource use (such as staff, equipment and overheads) (Healthcare Pricing Office 2022). The first step in the assignment of a discharge to a DRG is the classification of discharges by Major Diagnostic Category (MDC). There are 23 MDCs reflecting major systems of the body, e.g., MDC 4 represents diseases of the respiratory system. Within MDC, cases are further partitioned into surgical, medical and other categories[17], and by complexity (Bane 2015). In 2019, there were 807 DRGs in use (Healthcare Pricing Office 2020). Costs for each DRG are available from the price lists set by the Health Service Executive (HSE) as part of the Activity-Based Funding (ABF) initiative (see below for further information).

b. Analytic Sample

The HPO provided the research team with a datafile of all HIPE discharges over the period 2016-2021 that included the following ICD-10-AM chapters as either a principal or secondary diagnosis:

- Diseases of the circulatory system (I00 – I99)
- Diseases of the respiratory system (J00 – J99)
- Exposure to air pollution (Z58.1)

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[16] Use of ICD-10-AM, ACHI and ACS is complemented by the Irish Coding Standards (ICS); these are revised as required to reflect changing clinical practice and to ensure the classification and its application are relevant to the Irish healthcare system (Healthcare Pricing Office 2022).

[17] The DRGs are identified by a 4-character code. The first character in the code is alphabetic and refers to the MDC that the DRG belongs to. The second and third characters are numeric and identify whether the DRG is surgical, medical or other. The fourth character identifies the complexity level associated with the DRG (Bane 2015).

We focus on these diagnostic categories as the evidence base for causal impacts of air pollution on circulatory and respiratory disease is most robust (Anderson et al. 2003; Brook et al. 2010; Murray et al. 2020; OECD 2020). In ICD-10-AM, there is a specific diagnostic code for 'exposure to air pollution', although it is rarely used and no discharges were assigned this code as a principal diagnosis. Table A1 in the Appendix details the full list of variables available on the HIPE datafile provided to the research team that are used in this analysis. Due to the sensitivity of the data, information on many variables is either presented in aggregated form (age) or not available (county of residence). We analyse the data over the full four-year period in order to have sufficient sample size across some variables (e.g., hospitals).

We make a number of adjustments to the datafile in order to derive our sample for analysis. First, we focus only on discharges with a principal diagnosis of circulatory or respiratory disease. Second, we exclude the last two years (2020 and 2021), as public hospital activity was severely affected by the COVID-19 pandemic in these years. This exclusion also ensures that a consistent clinical coding system (8th revision of ICD-10-AM) is used for the sample period 2016-2019.[18] Third, we focus on emergency in-patient discharges only, i.e., those admitted for elective care as a day patient, or for maternity care, are excluded. Fourth, we exclude HIPE discharges with a length of stay (LOS) of over 365 days, and those in hospitals with very few circulatory or respiratory discharges.[19] Finally, we exclude a very small number of observations with missing information on key variables of interest (e.g., medical card status). This results in a final sample size of 464,639 discharges over the four-year sample period 2016-2019. To put these figures in context, this represents just over 25 per cent of all emergency inpatient discharges over the period 2016-2019 (Healthcare Pricing Office 2020).

Figure 1 illustrates how the distribution of discharges by principal diagnosis code varies over the sample period. Over the period 2016-2019, discharges with a primary diagnosis of circulatory disease accounted for 41 per cent of the total, and respiratory discharges 59 per cent. As noted, no discharges had a principal diagnosis of related specifically to air pollution (Z58.1).[20]

[18] From 1st January 2020, the coding system was updated to the 10th revision.

[19] We exclude hospitals in which the number of discharges is less than the 1 percentile number of observations per hospital over the period 2016-2019 – these are likely to be specialised hospitals (e.g., orthopaedics, maternity) which would not typically treat circulatory or respiratory patients.

[20] Just 14 discharges over the entire four-year period had a secondary diagnosis of Z58.1.

[insert Figure 1 here]

Table A2 in the Appendix presents additional clinical information on the analytic sample. Most discharges occurred in the winter months, particularly for respiratory discharges. For example, 42 per cent of respiratory discharges occurred in the four months November, December, January and February. The Charlson co-morbidity index is a score based on all secondary diagnoses (using ICD codes) that predicts the risk of death within 1 year of hospitalisation (Charlson et al. 1987). On average, discharges in our sample had a score of 1.2 on the Charlson comorbidity index (range 0-17). The average length of stay for circulatory discharges was 7.9 days, and 6.7 days for respiratory discharges.[21] Discharges were spread widely across HIPE hospitals and hospital groups, including the three paediatric hospitals (see Figures A1 and A2 in the Appendix). Although not presented in Table A2, the data indicate that most common principal diagnosis among the circulatory diagnoses was I50 (heart failure), accounting for 13 per cent of circulatory primary diagnoses. The most common principal diagnosis among the respiratory diagnoses was J22 (unspecified acute lower respiratory infection), accounting for 21 per cent of respiratory primary diagnoses. The median number of additional diagnoses was 3 for discharges with a principal diagnosis of circulatory disease, and 2 for discharges with a principal diagnosis of respiratory disease.

Table A3 in the Appendix presents data on the demographic and socioeconomic characteristics of the discharges included in our sample over the period 2016-2019. The data in Table A3 illustrate that 55 per cent of circulatory and respiratory discharges were aged 65+ (with discharges for respiratory disease more concentrated in younger age groups) and 46 per cent were female. As medical cards are primarily allocated on the basis of an income means test, medical card status can act as a useful proxy for socioeconomic status; in our sample, 67 per cent of discharges had a medical card. Similarly, public/private status refers to whether the patient saw the consultant on a public or private basis. It does not relate to the type of bed occupied nor is it an indicator of private health insurance (Keegan et al. 2020). Over the period 2016-2019, 85 per cent of discharges were classified as public.

[21] In 2018, the length of stay assigned for same day inpatients changed from one bed day to 0.5 bed days. This is based on an analysis of hospital data which shows that, on average, 0.5 days is a more appropriate measure of length of stay for this cohort of patients. Therefore, caution must be taken if comparing the average length of stay data before and after 2018 (Healthcare Pricing Office 2020, 201).

c. Identification of Discharges due to Air Pollution

With the exception of one dedicated ICD-10-AM code (Z58.1 'exposure to air pollution')[22], HIPE does not determine whether a discharge occurred due to (or was exacerbated by) exposure to air pollution. We use a variety of sources to estimate, for principal diagnoses in the circulatory and respiratory disease chapters, the proportion of discharges that may be attributed to air pollution. We gather data on estimates of the population attributable fraction (PAF), i.e., the proportion of total cases that can be attributed to air pollution.[23] As data on PAFs for (emergency) hospitalisations related to air pollution are not widely available, we rely instead on data on PAFs for morbidity due to ambient air pollution. Most estimates focus on attributable cases from ambient fine particulate matter ($PM_{2.5}$).

The main source is from work for the European Environment Agency (EEA) that estimates the morbidity-related health burden associated with exposure to three key air pollutants: fine particulate matter ($PM_{2.5}$), nitrogen dioxide (NO_2) and ozone (O_3) for 41 European countries in 2019 (Kienzler et al. 2022). Ten risk-outcome pairs, for which evidence for robust causal relationships were available, were considered by the EEA.[24] In this report, we use the data for Ireland from seven of those risk-outcome pairs that relate to circulatory and respiratory disease. We also use the high and low bounds to reflect uncertainty in the underlying assumptions.[25] For each risk-outcome pair, we identify the relevant sample of discharges using the ICD-10-AM codes (and apply any relevant age restrictions). The seven risk-outcome pairs examined in this paper are documented in Table 1 below.

[22] There are a series of ICD codes to account for conditions 'caused' by problems related to the physical environment (e.g., air, noise, soil pollution, etc.). A US study found that there were 341 patients in Florida who visited an ED due to environmental pollution exposure from 2016 to 2019 and 159 patients in Florida who were hospitalized (Ryan 2022). Patients exposed to air pollution frequently were diagnosed with asthma or other chronic obstructive pulmonary disease.

[23] The population attributable fraction represents the share of the environmental burden of disease attributable to the relevant environmental exposure (e.g., $PM_{2.5}$). It is a function of the relative risk (RR) associated with a particular exposure and is expressed as: $PAF = \frac{(RR-1)}{RR}$.

[24] Other work by the same consortium for the EEA calculated the all-cause mortality impact of $PM_{2.5}$, NO_2 and O_3 exposure for 41 European countries in 2019 (González Ortiz et al. 2021), and in 2020 (Soares et al. 2022). However, studies increasingly show that ambient air pollution is not only associated with mortality but also with morbidity due to several short-term and chronic conditions (Kienzler et al. 2022).

[25] Similar estimates are available from the Global Burden of Disease (GBD) study for 2019 (Murray et al. 2020). The risk-outcome pairs are slightly different to those used in the EEA analysis. Another potential source of data is from a recent report that assessed the burden of $PM_{2.5}$ air pollution on the island of Ireland, using updated estimates of relative risk to those used in the 2019 GBD calculations (Goodman et al. 2023). The report focused on deaths due to circulatory disease, and estimated that 7.6 per cent of premature deaths in Ireland and 8.6 per cent of deaths in Northern Ireland could be attributed to $PM_{2.5}$ in 2019.

[insert Table 1 here]

d. Assessment of Resource Use of Discharges due to Air Pollution

For discharges in each risk-outcome pair, we assess resource use using two metrics: bed days and DRG costs. Bed days are defined as the sum of length of stay for all discharges in each risk-outcome pair, as follows:

$$bed\ days_{r,o} = \left(\sum_{i=1}^{n} LOS_i\right) \times\ PAF_{r,o}$$

where LOS_i represents the length of stay for each discharge i in risk-outcome pair (r, o). For each risk-outcome pair, the total is then scaled by the PAF parameters presented in Table 1 (i.e., the share of morbidity attributable to air pollution) to derive mean, and lower and upper bounds, for total bed days.

In order to assess the costs of discharges due to air pollution, we use data from the ABF system for 2021. The ABF system determines funding to hospitals based on the number and mix of patients that they treat. In an ABF scheme a hospital receives a payment for each patient encounter. The exact payment for a given encounter depends on the type and complexity of the individual case (Bane 2015).[26] As the cost of care in emergency departments (EDs) is currently not included in the ABF system, and as all emergency inpatients will have been processed via an ED, and thereby incur hospital costs prior to their inpatient hospital stay, we apportion the average ED cost (€298 per ED attendance in 2018) (Keegan et al. 2020)[27] to all air pollution-related hospitalisations in this analysis. Total costs for all discharges in each risk-outcome pair are calculated as follows:

$$total\ costs_{r,o} = \left(\sum_{i=1}^{n} ABF_i + ED_i\right) \times PAF_{r,o}$$

where $ABF_i + ED_i$ represents the ABF and ED cost for each discharge in each risk-outcome pair (r, o). For each risk-outcome pair, total costs are then scaled by the PAF parameters presented in Table 1 (i.e.,

[26] ABF is currently in operation at 43 hospitals across Ireland, which account for approximately 90 per cent of national acute hospital activity. However, due to the COVID-19 pandemic, the previous system of block-grant funding of hospitals was re-introduced for 2021, 2022 and 2023 (Healthcare Pricing Office 2023).

[27] Expressed in 2021 prices using the CPI (€307).

the share of morbidity attributable to air pollution) to derive mean, and lower and upper bounds, for total costs.

3. Results

The data in Tables A2 and A3 presented a broad overview of the key clinical, demographic and socioeconomic characteristics of all emergency discharges with a principal diagnosis of circulatory or respiratory disease over the period 2016-2019. In this section, we focus in greater detail on the diagnoses identified in the EEA analysis, quantifying the resource utilisation (bed days and cost) associated with each of these diagnosis groups in turn. Before doing so, in Table 2, we present some key information on the clinical, demographic and socio-economic characteristics of the discharges included in each of the seven risk-outcome pairs. Not surprisingly, the largest number of discharges relates to the risk-outcome pair O_3 and respiratory hospitalisations for those aged 65+, as this risk-outcome pair encompasses the entire respiratory diagnosis ICD-10-AM chapter (J00-J99). On average, length of stay is substantially lower for children hospitalised for asthma than it is for adults. The longest average length of stay relates to hospitalisations for adults aged 25+ for diagnoses related to stroke. Hospitalisations for respiratory diagnoses such as COPD are concentrated among the older population, with a consequent higher average comorbidity score.

[insert Table 2 here]

In Figure 2, we show how discharges in each of the risk-outcome pairs are distributed across quarters of the year, and hospital groups.[28] Note that the data refer to all discharges for each risk-outcome pair, as individual discharges cannot be attributed air pollution. The data show that for the circulatory risk-outcome pairs (IHD and stroke), there is little variation across quarters of the year. However, the number of discharges for these conditions varies substantially across hospital groups. While some of this variation may be due to differences in air pollution exposure among those attending these hospitals, it is likely that this variation is also due to the general size and catchment area of different hospitals. For COPD discharges, and respiratory discharges for those aged 65+, the seasonal variation is much more apparent, with higher numbers of discharges in Q1 and Q4. Asthma discharges (for both

[28] See Table 1.1 in (Healthcare Pricing Office 2022) for a list of hospitals and corresponding hospital groups in Ireland.

children and adults) are distributed relatively evenly across the year, although there is evidence of higher admissions for both children and adults in Q4 (October – December).

[insert Figure 2 here]

Table 3 shows how resource use (total bed days, total DRG costs) vary across the seven risk-outcome pairs examined. The most resource-intensive risk-outcome pairs are both related to $PM_{2.5}$ air pollution. Hospitalisations for COPD that can be attributed to $PM_{2.5}$ air pollution accounted for an average of 36,776 bed days, and cost an average of €29.3m over the period 2016-2019. Hospitalisations for stroke that can be attributed to $PM_{2.5}$ air pollution accounted for an average of 21,759 bed days, and cost an average of €20.1m over the period 2016-2019. Other risk-outcome pairs (e.g., hospitalisations for asthma for children aged 0-14) account for much smaller shares of total resources. For the five risk-outcome pairs that contain mutually exclusive ICD-10-AM diagnosis codes, total bed days for conditions related to air pollution accounted for 63,572 bed days (range 17,767-115,996) over the period 2016-2019.[29] In terms of total costs, the costs ranged from €15.0m to €105.8m, with an average total cost of €56.0m. To put these figures in context, in 2019, total expenditure on acute hospital services in Ireland (emergency and elective inpatient, as well as day- and outpatient, care) amounted to €6.8bn (Health Service Executive 2020), and total emergency inpatient bed days amounted to 2.8m (Healthcare Pricing Office 2020).

The variation in resource use across risk-outcome pairs reflects differences in the number of discharges, the attributable share, LOS, DRGs and associated ABF costs for each risk-outcome pair. For bed days, resource use for each risk-outcome pair is a function of the attributable share, the number of discharges and the average LOS. For example, while a similar share of hospitalisations for asthma for children aged 0-14 and hospitalisations for COPD for adults aged 25+ are estimated to be attributable to $PM_{2.5}$ air pollution (see Table 2), the combination of shorter average LOS and fewer discharges means that the total bed days for asthma hospitalisations for children aged 0-14 are considerably less than for COPD discharges for adults aged 25+. Similarly, the higher share of asthma hospitalisations attributable to air pollution among children aged 0-14 (compared with adults aged 25+) means that total costs for the treatment of hospitalisations for asthma for children aged 0-14 and

[29] To put these figures in context, total emergency inpatient bed days amounted to 2.8m in 2019 alone (Healthcare Pricing Office 2020).

for adults aged 25+ are broadly similar, despite the fact that fewer children are hospitalised for asthma overall than adults.

[insert Table 3 here]

4. Discussion

In comparison with other European countries, levels of ambient air pollution in Ireland are amongst the lowest; for example, in 2017, Ireland had the sixth lowest annual concentration of $PM_{2.5}$ of 37 European countries (European Environment Agency 2019). However, the WHO note that there is no safe level of air pollution (WHO 2021), and there are concerns over exceedances of various pollutants at local level in Ireland, in particular after the revision of WHO ACQs in 2021 (EPA 2022). While there have been some assessments of the mortality burden associated with ambient air pollution in Ireland (European Environment Agency 2019, 2022; Goodman et al. 2023)[30], there is a lack of evidence on the broader healthcare costs associated with ambient air pollution in Ireland.

In this paper, we estimated the healthcare costs of air pollution in Ireland, using data on emergency inpatient hospital admissions and costs over the period 2016-2019 from the Hospital Inpatient Enquiry (HIPE) system, supplemented with data on population attributable fractions for specific conditions with causal links to air pollution (e.g., asthma). The next phase of this research project will incorporate non-acute healthcare costs associated with air pollution in Ireland. The results indicate that the most resource-intensive risk-outcome pairs are both related to $PM_{2.5}$ air pollution. Hospitalisations for COPD that can be attributed to $PM_{2.5}$ air pollution accounted for an average of 36,776 bed days, and cost an average of €29.3m over the period 2016-2019. Hospitalisations for stroke that can be attributed to $PM_{2.5}$ air pollution accounted for an average of 21,759 bed days, and cost an average of €20.1m over the period 2016-2019. Other risk-outcome pairs (e.g., hospitalisations for asthma for children aged 0-14) accounted for much smaller shares of total resources. For the five risk-outcome pairs that contained mutually exclusive ICD-10-AM diagnosis codes, total bed days for conditions related to air

[30] Using data for 2019, (Goodman et al. 2023) estimate that approximately 1,700 premature deaths (680 from cardiovascular disease) per annum in Ireland can attributable to exposure to $PM_{2.5}$. These premature mortality estimates for $PM_{2.5}$ are higher than those published by the European Environment Agency (EEA) or the Global Burden of Disease study which ranged from 535 to 1,300 (European Environment Agency 2019, 2022; Murray et al. 2020). This reflects the authors' use of updated dose response functions based on growing research evidence that exposure to $PM_{2.5}$ is more harmful than previously thought.

pollution accounted for 63,572 bed days (range 17,767-115,996) over the period 2016-2019.[31] In terms of total costs, the costs ranged from €15.0m to €105.8m, with an average total cost of €56.0m. To put these figures in context, in 2019, total expenditure on acute hospital services in Ireland (emergency and elective inpatient, as well as day- and outpatient, care) amounted to €6.8bn (Health Service Executive 2020), and total emergency inpatient bed days amounted to 2.8m (Healthcare Pricing Office 2020).

Additionally, the costs of treating stroke cases attributed to NO_2 pollution amounted to €9.4m (range €4.9m – €13.5m), and the costs of treating all respiratory hospitalisations among those aged 65+ attributable to O_3 air pollution was €3.0m (range €0.5m - €5.6m). A previous analysis by the OECD estimated that the healthcare costs of treating lung cancer, circulatory disease and respiratory diseases due to ambient exposure to $PM_{2.5}$ and O_3 in 2017 amounted to €0.15bn (OECD 2016, 2020). In England, it was estimated that the total health and social care costs (i.e., acute as well as non-acute costs) of $PM_{2.5}$ in 2017 were £41.2m (rising to £76.1m when diseases were included for which the evidence base was less well established) (Pimpin et al. 2018; Public Health England 2018).

As with any analysis of this type, there are inevitable strengths and limitations. As the evidence base for harmful effects of air pollution is most robust for circulatory and respiratory diagnoses, the analysis in this paper focused on these conditions. However, there is an emerging evidence base for harmful effects of ambient air pollution on other aspects of physical and mental health (see Section 1), and that could be incorporated in future analyses. In addition, the current analysis focused only on emergency inpatient hospitalisations in public hospitals, which necessarily omits other hospitalisations (e.g., outpatient visits, attendances in private hospitals) which may also be attributable to air pollution. The lack of a unique patient identifier in HIPE means that we cannot examine the extent to which circulatory and respiratory diagnoses are concentrated among a cohort of patients who repeatedly attend hospital, or are more widely distributed.[32] With data such as HIPE, it is not possible to distinguish the short- and long-term effects of exposure to air pollution on emergency inpatient hospitalisations. For instance, for a diagnosis such as asthma, symptoms can arise from

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[31] This equates to approximately 43 emergency inpatient hospital beds per annum for air pollution-related discharges. This resource use occurs in the context of a healthcare system with severe capacity constraints. Overall, it is estimated that 300 extra hospital beds per annum are required to keep up with demand and demographic pressures (McQuinn et al. 2023).

[32] In a study of patients admitted as emergency inpatients using HIPE data for one major teaching hospital in Dublin over a 13-year period, (Cournane et al. 2016) found that just under 30 per cent of the respiratory patients were admitted at least five times, while approximately 10 per cent were admitted 10 times.

contemporaneous exposure (in as quickly as one hour of exposure), from cumulative exposure over several days, or several days after exposure (Neidell 2009). In addition, if people respond to higher pollution levels by increasing avoidance behaviour, then the estimated effect of pollution on health and healthcare utilisation can be biased downwards, and the full cost of pollution exposure will therefore be underestimated (Moretti and Neidell 2011).[33] Finally, while we use PAF estimates from the EEA to estimate the share of discharges within each risk-outcome pair attributable to the various air pollutants (and use ranges to account for uncertainty), there is a wide variety of PAF estimates in the available literature.[34] While there is now a dedicated ICD-10-AM code for 'exposure to air pollution', just 14 discharges over the period 2016-2019 had a secondary diagnosis of 'exposure to air pollution' in Ireland. (Ryan 2022) discuss the many reasons why this code may be underutilised by clinicians (e.g., the clinician would need to know about and document the patient's pollution exposure, and about the pathways linking air pollution and health).

Despite these limitations, the approach taken in this paper, using hospital discharge and cost data to identify the resource costs associated with air pollution, circumvents the need for high-frequency monitoring data on air pollution that can be linked to hospitalisations. Most of these studies using ambient air pollution concentration levels, which may not be a good indicator of personal exposure.[35] Personal exposure is influenced by the different microenvironments or activities an individual experiences (e.g., time in traffic, indoor sources, second-hand tobacco smoke, occupational exposure, and degree of penetration of ambient air pollution into homes, etc.) (Brook et al. 2010).

In terms of the policy response, the recent Clean Air Strategy commits to achieving the final WHO ACQ values by 2040 (Government of Ireland 2023), and at EU level, the proposed revision to the Ambient Air Quality Directive will set interim 2030 EU air quality standards, aligned more closely with WHO guidelines, and set Europe on a trajectory to achieve zero pollution for air by 2050. Achieving these targets will require continued policy focus on measures such as moving away from burning of solid fuels. In addition, policy measures to mitigate the impacts of climate change, such as decarbonising home heating, promoting active travel and transitioning to electric vehicles, will have concomitant

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[33] Indeed, (Neidell 2004, 2009) show how individuals respond to 'smog alerts' in California by reducing outdoor interactions, with the result that the estimated effects of O_3 concentrations on hospitalisations for children and older people in particular are biased downwards.

[34] See Box 2.5 in (OECD 2020) for a good discussion of the implications of differences in PAF estimates for differences in estimates of the mortality burden of air pollution across Europe.

[35] See (Lleras-Muney 2010) for a good discussion of the difficulties in attributing air pollution exposure to individuals using data from pollution monitors.

benefits for air quality and population health (van Daalen et al. 2022). In addition, those in more disadvantaged social positions may be more likely to live in areas that are exposed to harmful environmental conditions, or to work in jobs that involve increased environmental risks (European Environment Agency, 2018). Certain population groups (e.g., children, older people, those in more disadvantaged social positions, etc.) may also be more vulnerable to the health-damaging effects of environmental exposures such as air pollution, due to other characteristics such as poor housing conditions, chronic disease, etc. (European Environment Agency, 2018; OECD, 2020). In this context, further policy measures will be required in order to ensure that the targeted reductions in air pollution benefit those who are most exposed and/or most vulnerable to its effects.

Tables and Figures

Figure 1 Number of Discharges by Primary Diagnosis, 2016-2019

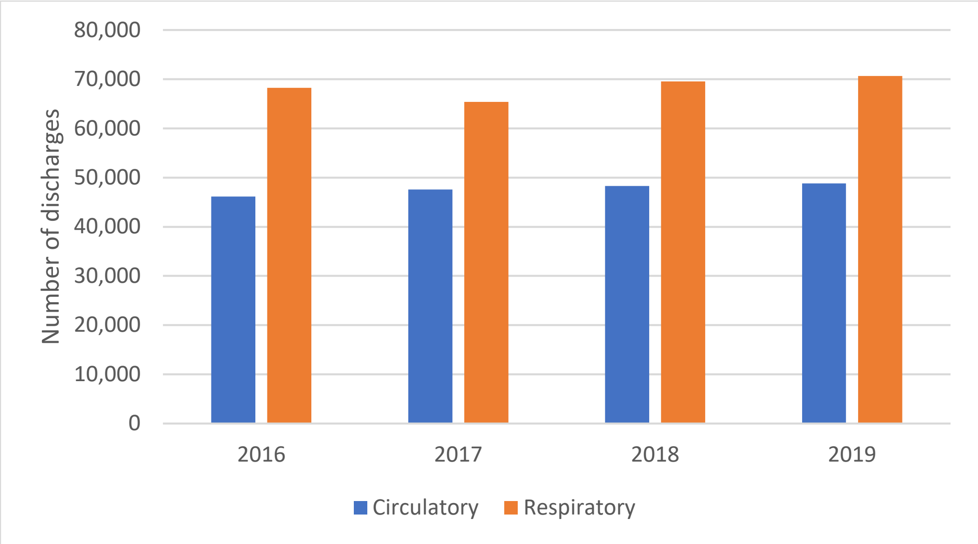


Table 1 Population Attributable Fractions for Air Pollution Risk-Outcome Pairs

Risk-outcome pair	ICD-10-AM code	Population attributable fraction[1]		
		lower bound	**mean**	**upper bound**
1. $PM_{2.5}$ Asthma (aged 0-14)	J45-J46	0.026	0.075	0.119
2. $PM_{2.5}$ COPD (aged 25+)	J40-44; J47	0.031	0.079	0.124
3. $PM_{2.5}$ IHD (aged 25+)	I20-I25	0.000	0.011	0.049
4. $PM_{2.5}$ Stroke (aged 25+)	I60-69	0.005	0.049	0.095
5. NO_2 Asthma (aged 15+)	J45-46	0.028	0.044	0.061
6. NO_2 Stroke (aged 25+)	I60-69	0.012	0.023	0.033
		Attributable hospital cases[2]		
		lower bound	**mean**	**upper bound**
7. O_3 Respiratory (aged 65+)	J00-J99	2	14	25

Notes:

[1] The population attribution fraction represents the share of the environmental burden of disease attributable to the respective risk factor (e.g., $PM_{2.5}$).

[2] Unlike the other risk-outcome pairs, a relative risk estimate is not available. This means that the PAF could be calculated for this risk-outcome pair. Alternatively, attributable hospital admission cases were calculated.

Source: (Kienzler et al. 2022)

Table 2 Emergency Inpatient Hospital Clinical and Demographic Characteristics for Conditions related to Air Pollution, 2016-2019

Risk-outcome pair	ICD-10-AM code	Total number of discharges	Average length of stay	% aged 0-14	% aged 65+	% male	Average Charlson morbidity score
1. $PM_{2.5}$ Asthma (aged 0-14)	J45-J46	6,112	1.8	100.0	0.0	62.4	1.0
2. $PM_{2.5}$ COPD (aged 25+)	J40-44; J47	60,975	7.6	0.0	75.3	48.9	1.7
3. $PM_{2.5}$ IHD (aged 25+)	I20-I25	46,448	5.7	0.0	57.6	70.4	1.2
4. $PM_{2.5}$ Stroke (aged 25+)	I60-69	28,276	15.7	0.0	71.4	55.6	2.3
5. NO_2 Asthma (aged 15+)	J45-46	9,688	3.1	0.0	22.3	33.3	1.2
6. NO_2 Stroke (aged 25+)	J45-46	28,276	15.7	0.0	71.4	55.6	2.3
7. O_3 Respiratory (aged 65+)	J00-J99	137,087	9.6	0.0	100.0	50.7	1.6

Notes:

[1] As PAFs are not taken into account yet, the clinical and demographic/socioeconomic information for the two risk-outcome pairs related to stroke is identical.

Table 3 Emergency Inpatient Hospital Resource Use for Conditions related to Air Pollution, 2016-2019

Risk-outcome pair	**ICD-10-AM code**	**Bed days attributable to air pollution**			**DRG cost attributable to air pollution (€)**		
		lower bound	**mean**	**upper bound**	**lower bound**	**mean**	**upper bound**
1. $PM_{2.5}$ Asthma (aged 0-14)	J45-J46	281	811	1,287	492,709	1,421,277	2,255,093
2. $PM_{2.5}$ COPD (aged 25+)	J40-44; J47	14,431	36,776	57,725	11,498,105	29,301,622	45,992,420
3. $PM_{2.5}$ IHD (aged 25+)	I20-I25	0	2,914	12,980	0	3,727,667	16,605,063
4. $PM_{2.5}$ Stroke (aged 25+)	I60-69	2,220	21,759	42,186	2,046,870	20,059,322	38,890,523
5. NO_2 Asthma (aged 15+)	J45-46	835	1,312	1,819	925,149	1,453,805	2,015,502
Total		**17,767**	**63,572**	**115,996**	**14,962,833**	**55,963,694**	**105,758,601**
6. NO_2 Stroke (aged 25+)	J45-46	5,329	10,214	14,654	4,912,487	9,412,487	13,509,340
7. O_3 Respiratory (aged 65+)	J00-J99	575	3,567	6,751	479,345	2,971,939	5,624,315

Notes:

[1]See Table 1 for data on the population attributable fraction (or attributable hospital cases) for each risk-outcome pair.

[2] In order to focus on discharges with unique ICD-10-AM codes, the total includes the first five risk-outcome pairs only (i.e., we cannot add NO_2 stroke estimates to the total as these ICD-10-AM codes are already used in the estimates for $PM_{2.5}$ stroke).

Figure 2 Distribution of discharges for each risk-outcome pair by hospital group[36] and quarter of the year

a) $PM_{2.5}$

Asthma (aged 0-14)

	Q1	Q2	Q3	Q4	% discharges
Ireland East	153	143	119	196	10.0
Dublin Midlands	47	60	47	87	3.9
RCSI Group	97	119	96	133	7.3
Childrens Group	398	454	406	528	29.2
South Southwest	249	244	235	311	17.0
UL Group	115	74	78	96	5.9
Saolta	362	382	413	470	26.6

b) $PM_{2.5}$

COPD (aged 25+)

	Q1	Q2	Q3	Q4	% total discharges
Ireland East	3,941	3,479	3,032	4,070	23.8
Dublin Midlands	3,220	2,397	2,184	2,865	17.5
RCSI Group	2,565	2,213	1,869	2,594	15.2
South Southwest	3,087	2,390	2,173	2,763	17.1
UL Group	1,680	1,348	1,186	1,516	9.4
Saolta	3,067	2,361	2,184	2,791	17.1

c) $PM_{2.5}$

IHD (aged 25+)

	Q1	Q2	Q3	Q4	% total discharges
Ireland East	2,613	2,641	2,574	2,699	22.7
Dublin Midlands	2,415	2,270	2,414	2,320	20.3
RCSI Group	1,449	1,549	1,499	1,405	12.7
South Southwest	2,010	2,114	2,171	1,971	17.8
UL Group	1,000	921	940	884	8.1
Saolta	2,166	2,195	2,158	2,070	18.5

[36] See Table 1.1 in (Healthcare Pricing Office 2022) for a list of hospitals and corresponding hospital groups in Ireland.

d) $PM_{2.5}$ /NO_2

Stroke (aged 25+)

	Q1	Q2	Q3	Q4	% total discharges
Ireland East	1,484	1,465	1,478	1,570	21.2
Dublin Midlands	1,028	1,064	940	1,041	14.4
RCSI Group	1,676	1,636	1,659	1,710	23.6
South Southwest	1,402	1,316	1,380	1,360	19.3
UL Group	501	468	447	475	6.7
Saolta	1,029	1,017	1,080	1,050	14.8

e) $PM_{2.5}$

Asthma (aged 15+)

	Q1	Q2	Q3	Q4	% total discharges
Ireland East	533	495	457	609	21.6
Dublin Midlands	386	330	327	477	15.7
RCSI Group	421	392	402	520	17.9
Childrens Group	6	5	8	7	0.3
South Southwest	445	396	368	482	17.5
UL Group	296	217	240	293	10.8
Saolta	389	388	378	421	16.3

f) O_3

Respiratory hospitalisations (aged 65+)

	Q1	Q2	Q3	Q4	% total discharges
Ireland East	9,198	7,305	6,174	8,767	22.9
Dublin Midlands	6,428	4,854	4,199	5,817	15.5
RCSI Group	6,296	5,020	4,281	6,041	15.8
South Southwest	7,584	5,627	4,809	6,612	18.0
UL Group	3,533	2,684	2,222	2,991	8.3
Saolta	7,872	6,168	5,509	7,096	19.4

Table A1 HIPE Variable Definitions

Variable	Definition
Clinical information	
Month of admission	January - December
Year of admission	2016, 2017, 2018, 2019
Type of admission	Emergency, emergency readmission, newborn
Diagnosis code for principal diagnosis[1]	ICD-10-AM code
Comorbidity	Charlson comorbidity score[2]
Hospital Group	Broad hospital group (e.g., Ireland East, Dublin Midlands, etc.)
Length of stay	Length of inpatient hospital stay in days
DRG	Diagnosis-related group
ABF cost	ABF cost of DRG (2021 prices)
Demographic/socioeconomic information	
Age	Age in ten-year age bands, up to 85+[3]
Sex	Male, female
Medical card	Medical card status (yes, no)
Public patient	Public consultant (yes, no)

Notes:

[1] ICD-10-AM codes are also available for up to 29 secondary diagnoses

[2] The Charlson co-morbidity index is a score based on all of a patient's diagnoses (using ICD codes) that predicts the risk of death within 1 year of hospitalisation (Charlson et al. 1987).

[3] Children aged under 15 are grouped into one composite group, aged 0-14.

Table A2 Clinical Characteristics of Full Analytic Sample[2]

	Circulatory Disease[1] (ICD: I00-I99)	**Respiratory Disease[1]** (ICD: J00-J99)	**All Circulatory and Respiratory Diseases[1]**
Month of admission			
January	8.5	11.6	10.4
February	7.8	9.0	8.5
March	8.7	8.9	8.8
April	8.3	7.9	8.1
May	8.7	7.6	8.1
June	8.1	6.7	7.3
July	8.4	6.3	7.1
August	8.5	5.8	6.9
September	8.2	7.0	7.5
October	8.5	7.9	8.1
November	8.3	9.1	8.8
December	8.0	12.3	10.5
Year of admission			
2016	24.2	24.9	24.6
2017	24.9	23.9	24.3
2018	25.3	25.4	25.4
2019	25.6	25.8	25.7
Hospital Group			
Ireland East	22.8	20.8	21.6
Dublin Midlands	16.3	13.6	14.7

RCSI	16.1	14.7	15.3
Children	0.4	5.9	3.6
South Southwest	18.4	17.8	18.0
UL	7.9	7.9	7.9
Saolta	18.0	19.4	18.8
Type of admission			
Emergency	98.9	98.1	98.4
Emergency readmission	1.1	1.4	1.3
Newborn	0.1	0.5	0.3
Comorbidity (Charlson score)	1.3	1.1	1.2
Length of stay (days)	7.9	6.7	7.2

Notes:

[1] Refers to principal diagnosis based on ICD-10-AM codes

[2] With the exception of length of stay (which is presented as average number of days) and comorbidity (which is presented as average Charlson score), the data here refer to the percentage of total discharges in each group.

Table A3 Demographic and Socioeconomic Information of Full Analytic Sample[2]

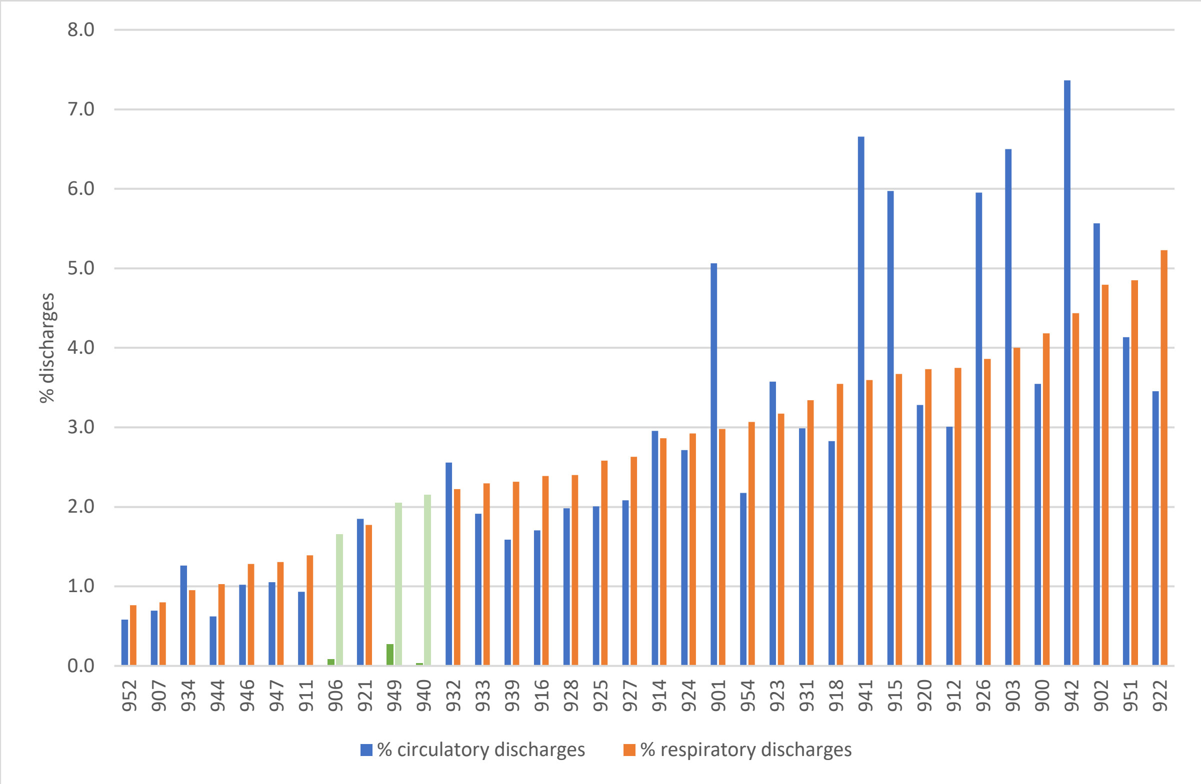
	Circulatory Disease[1] I00-I99	**Respiratory Disease[1]** J00-J99	**All Circulatory and Respiratory Diseases[1]**
Age Category			
Age 0-14	2.2	20.2	12.8
Age 15-24	1.3	4.2	3.0
Age 25-34	2.0	3.6	2.9
Age 35-44	4.8	5.0	4.9
Age 45-54	9.9	6.3	7.8
Age 55-64	16.7	10.7	13.2
Age 65-74	24.6	18.5	21.0
Age 75-84	25.5	19.9	22.2
Age 85+	13.1	11.7	12.3
Sex			
Male	59.0	50.7	54.1
Female	41.1	49.3	45.9
Medical Card			
No	34.7	31.0	32.5
Yes	65.3	69.0	67.5
Public Patient			
No	83.1	85.9	84.8
Yes	16.9	14.1	15.3

Notes:

[1] Refers to principal diagnosis based on ICD-10-AM codes

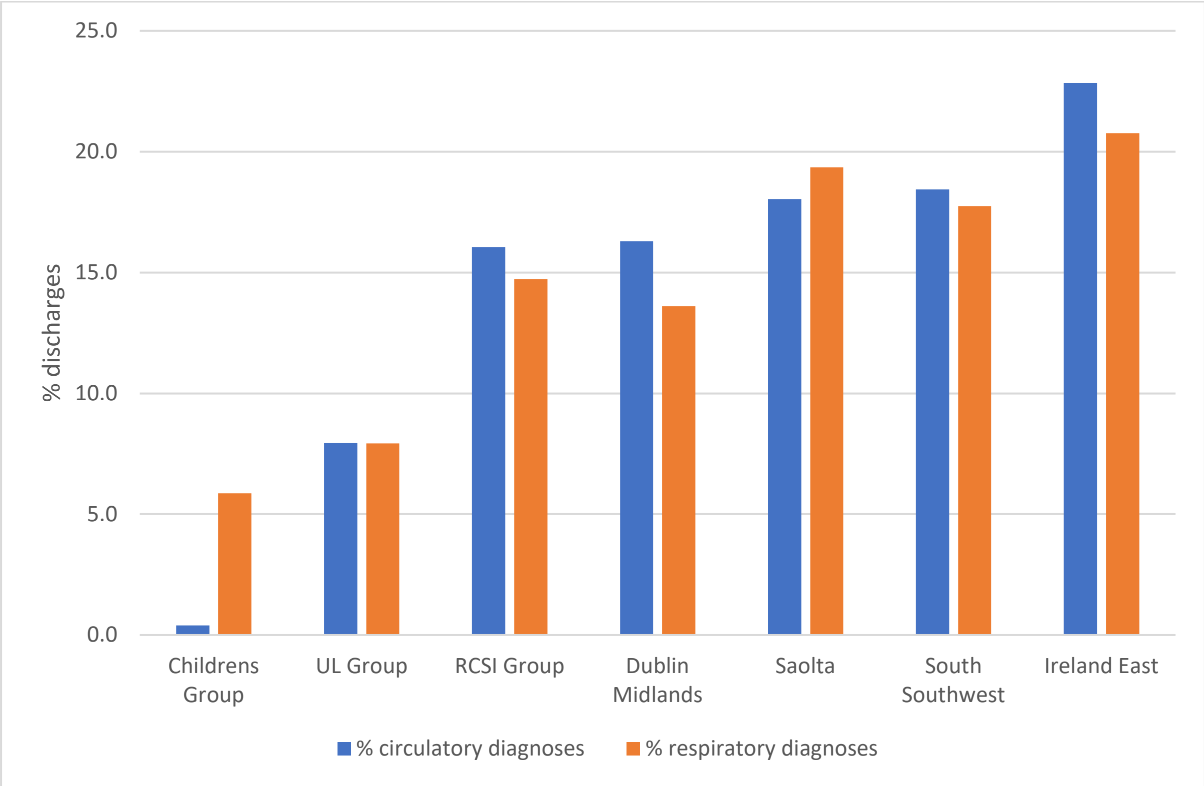
[2] The data here refer to the percentage of total discharges in each group.

Figure A1 Distribution of Circulatory and Respiratory Discharges by Hospital[1] (2016-2019)



Notes:
[1] Hospital codes are anonymised, although the hospitals marked in green indicate the three paediatric hospitals in Ireland (Crumlin, Tallaght, Temple St).

Figure A2 Distribution of Circulatory and Respiratory Discharges by Hospital Group (2016-2019)



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