

Improved Emissions Inventories for NO_x and Particulate Matter from Transport and Small Scale Combustion Installations in Ireland (ETASCI)

Part 2 of 2*

Authors:

Stephen Morrin, Dr. William J. Smith, Dr. David J. Timoney



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EPA Research Programme 2014-2020

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Small Scale Combustion Installations in
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1. Executive Summary

1.1 Project context

Under European law and the Convention on Long-Range Transboundary Air Pollution (CLRTAP), all countries that are a party to CLRTAP must submit an annual Emission Inventory Report, detailing quantities of air pollutants from a wide range of sectors. These quantities are estimated by combining activity data – detailing fuel use by type and sector – with emission factors (EFs), which quantify the mass of each pollutant emitted per unit energy used.

The Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (EMEP) was established under CLRTAP in 2001. CLRTAP, through EMEP, has implemented a number of directives and programmes, such as Clean Air For Europe (CAFE) and the Thematic Strategy on Air Pollution [1 2], with the aim of driving down emissions from specific sources.

EMEP is also responsible for compiling European-wide emission inventories, and for making emission projections. In this capacity, EMEP publishes a Guidebook listing emission factor (EF) values to be used by member states when estimating emission quantities, for all pollutants, across all sectors. However, it is acknowledged that significant uncertainty is associated with some of the EF values in the Guidebook, due primarily to gaps in the available data. A key objective of this project is the elimination of these data gaps for estimating emissions of particulate matter (PM) and oxides of nitrogen (NO_x) in Ireland.

1.2 The ETASCI project

The ETASCI project is divided into **two** complementary but distinct work packages (WPs) and final reports. **WP1** focuses on emissions from road transport and is detailed in EPA Research Report 148 – Part 1 of 2. **WP2** examines emissions from non-road transport and from small combustion installations (SCIs) and is detailed in EPA Research Report 149 – Part 2 of 2.

In this context, an SCI is defined as any stationary combustion installation with a thermal input rating of less than 50 MW. SCIs therefore encompass a huge range of appliances, from low-capacity boilers used to heat small houses, right up to large industrial units. Installations with a capacity greater than 50 MW fall under the Large Combustion Plant Directive [3], and are therefore very closely monitored and controlled.

1.3 Focus of this report

Although both WPs were intended to run in parallel, difficulties in recruiting a suitable PhD student for WP1 delayed its start by 20 months. This report, therefore, summarises WP2 activity only.

The two strands in WP2 – non-road transport, and stationary SCIs – are essentially independent of one another. They were examined separately in WP2, and are presented separately in this report. The non-road transport study uses emission factors (EFs) as specified by engine and vehicle manufacturers, but developed improved activity data from an in-depth analysis of

traffic volumes for air, rail and marine transport.

The SCI strand, conversely, focused on developing improved EFs that accurately represent Irish operational conditions. The paucity of disaggregated data for SCIs precluded significant refinement of the activity aspect of the model, and the authors warmly welcome the inclusion of relevant questions in the 2011 Census form. Within the SCI strand, the report focuses primarily on the residential heating market. This emphasis is justified on two main grounds: first, with almost 1.3 million homes in the Republic of Ireland having central heating [4], the sheer number of domestic SCIs far outweighs that of all other sectors combined; second, the technology used in combustion appliances is broadly independent of size, and so EFs measured for a domestic-scale appliance should be broadly in line with those of a larger unit. Field measurements carried out as part of WP2 tend to support this assertion.

Accurate measurement of PM emissions (and hence EFs) in the field is difficult, time-consuming, and requires specialised and expensive equipment. It was therefore decided to establish a laboratory test facility at UCD, in which domestic boilers would undergo simulated real-world operation, and wherein PM emissions could reliably be determined using less expensive (but very time-intensive) equipment. A selection of domestic boiler installations was monitored over a period of several months in order to establish a duty-cycle representative of real-world operation. These duty cycles were then imposed on boilers installed in the UCD test facility, and the corresponding EFs established. Particular attention was paid to

examining the effect of transient start/stop operation on the measured EFs. A field measurement campaign was carried out as a complement to the lab-based measurements, the aim of which was to investigate how EFs from real-world boilers of varying ages and service levels compared to those measured in the modern, state-of-the-art boilers in the laboratory. The accuracy of these field-measured EFs is considerably lower than the lab-based measurements, as it was not possible to measure or control a whole host of operating parameters.

Whilst oil- and gas-fired boilers dominate the residential heating sector in Ireland, recent years have seen a significant growth in the use of wood-pellet boilers. All three boiler types were therefore analysed in this study.

1.4 Principal findings from WP2

The results from WP2 represent a substantial stride along the road to developing a comprehensive, up-to-date, and accurate estimate of Irish NO_x and PM emissions.

Emissions from stationary SCIs, and from non-road transport, were investigated. It was found that, in general, the existing Environmental Protection Agency (EPA) approach tends to under-estimate NO_x but over-estimate PM emissions from non-road transport. However, as the absolute contribution of this sector to overall emission levels is small relative to that from SCIs (and from road transport), there is little to be gained from devoting additional resources to refining these estimates.

For stationary SCIs, in contrast, it was found that the emission factors (EFs) currently being used to compile emission inventory reports, significantly over-estimate the total NO_x and PM emissions from the heating sector. Moreover, although the PM emission factor for oil- and gas-fired boilers was found to be relatively independent of operating condition, PM emissions from the wood-pellet burner tested in the laboratory were seen to be highly sensitive to operating mode. The NO_x emission factor was insensitive to operating mode for each appliance.

Samples of the PM emitted from both the oil-fired and wood-pellet burners were examined using scanning electron microscopy (SEM), revealing substantial differences in PM size, shape, and composition between burner types. PM from the wood-pellet boiler consisted mainly of inorganic salt particles composed of potassium, sulphur, and chlorine. These salts had been released to the gas phase from ash elements in the fuel, and subsequently condensed to form regular, crystalline particles, some of which were quite large in size. Particles from oil combustion, on the other hand, had a different morphology and composition. Oil-derived PM were found to be almost perfectly spherical in shape, and existed either as individual particles on the collection surface, or as an agglomeration of spherical particles. They were composed mainly of carbon, oxygen and silicon.

1.5 Key recommendations from WP2

Based on the investigation carried out in this research project, there are a number of recommendations to be made.

1. Current emission inventories should be re-calculated using the newly measured emission factors (EFs). This will result in significant reductions in reported NO_x and PM emissions from the heating sector, and could be implemented quite easily.
2. The emissions measurement facility developed for this project should be leveraged. Valuable emissions-measuring capabilities and expertise have been developed through this project that can be maintained and developed at relatively low cost. A modest investment in additional equipment would significantly enhance the PM measurement proficiency, and expand the scope of the laboratory to include measurement of other important emissions, such as oxides of sulphur, volatile organic compounds, and PAH (Polycyclic Aromatic Hydrocarbons).
3. The fate of SCI-derived PM should be investigated. Domestic SCIs, in particular, generally emit pollutants into the air volumes used for human respiration. Exposure to these pollutants, and subsequent health impacts, are therefore highly dependent on local emission concentrations, which are a function not only of the emitted mass, but of the subsequent dispersion processes and PM aerodynamic characteristics.

4. The characteristics of SCI-derived and ambient PM should be further explored. Preliminary analysis of the Scanning Electron Microscopy (SEM) images clearly demonstrates that all PM is not equal – substantial variations in PM size, shape, and composition are evident. These characteristics, in turn, strongly influence the health and climate-change impacts of PM at local and global scales respectively. The linkage to the UCD Advanced

Imaging Facility offers significant potential to deepen the understanding of ambient, as well as of combustion-derived, PM.

PM and NO_x have significant adverse health impacts, such as increasing respiratory illness in the case of NO_x, while ambient levels of PM have been conclusively linked to increases in mortality rates. Inhaled PM can also clog airways in the lungs leading to pulmonary and respiratory illness.

2. Literature Review

2.1 CLRTAP and the history of emissions monitoring

Air pollution has been one of Europe's main political concerns since the 1970s. In the 1960s, scientists demonstrated the interrelationship between sulphur emissions in continental Europe and the acidification of Scandinavian lakes [5]. Between 1972 and 1977, several studies confirmed the hypothesis that air pollutants could travel several thousands of kilometres before deposition and damage occurred. This also implied that cooperation at the international level was necessary to solve problems, such as acidification, and, hence, in 1979, the Convention on Long-Range Transboundary Air Pollution (CLRTAP) [6] was signed by 34 Governments and the European Community (EC). The Convention was the first international legally binding instrument to deal with problems of air pollution on a broad regional basis. It laid down the general principles of international cooperation for air pollution abatement, as well as setting up an institutional framework bringing together research and policy. A comprehensive review of the first twenty-five years of CLRTAP is provided by Sliggers and Kakebeeke [7].

Under CLRTAP, the Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (EMEP) was established. Its main objective is to 'regularly provide governments and subsidiary bodies under the LRTAP Convention with qualified scientific information to support the development and further evaluation of the

international protocols on emission reductions negotiated within the Convention' [8]. As the source of information on the emission, transport and deposition of air pollution, EMEP has a major role to play in the establishment of air pollution protocols to the Convention. Under EMEP, a number of task forces were established, amongst them the Task Force on Measurements and Modelling (TFMM) and the Task Force on Integrated Assessment and Modelling (TFIAM). EMEP has three main strands – collection of emissions data, measurement of air and precipitation quality, and modelling of atmospheric transport and deposition. This research project focuses on the first strand: collection of emissions data, specifically that pertaining to NO_x and particulate matter (PM).

Clean Air for Europe (CAFE) was launched in March 2001 with Communication COM(2001)245 [1]. Its aim is to establish a long-term, integrated strategy to tackle air pollution and to protect against its effects on human health and the environment. As such, CAFE addresses both local air quality and long-range transboundary concerns. Its programme of technical analysis and policy development underpinned the development of the Thematic Strategy on Air Pollution, adopted by the European Commission on 21 September 2005 [2]. Again, EMEP is the primary source of information on the emission, transport and deposition of air pollution.

The overall approach adopted by the EU in pursuit of protecting air quality comprises multiple interconnected threads. Data from

the World Health Organisation (WHO) and from peer-reviewed studies is used to establish acceptable and/or desirable air quality standards. Monitoring programmes coordinated under EMEP determine the level to which these targets are attained. Computer models, such as RAINS [9] and GAINS [10], model the socio-economic and bio-physical impacts of transboundary air pollution on human health and ecosystems. Directives, such as the Large Combustion Plant Directive [11] and Integrated Pollution Prevention and Control [12], are used to drive down emissions from specific sources.

2.2 Ireland's energy use

According to their most recent report, focussing on 2009, the Sustainable Energy Authority of Ireland (SEAI) estimates that Ireland's total primary energy requirement (TPER) is 14.9 million tonnes of oil equivalent (MTOE) [13]. Out of this, the

residential heating sector accounted for 2.7 MTOE (18.3%), while the commercial heating sector used 0.9 MTOE (6.1%). Added to this is the non-road transport sector, accounting for 0.8 MTOE (5.5%), meaning that the sectors under examination in this report account for almost 30% of all of the energy used in this country.

According to the most recent census data available, there are just fewer than 1.5 million houses in the Republic of Ireland, with just 9% of these having no central heating [4]. Figure 2.1 shows the breakdown of residential fuel use for space heating in Ireland. The data was collected by the SEAI, by monitoring tariffs and sales of all of the relevant fuel types. As can be seen, there has been huge changes in the type of central heating used in Ireland over the last couple of decades. In 1987, almost half the homes in Ireland did not have any form of central heating, whereas by 2005, this had reduced to less than one in 10 homes.

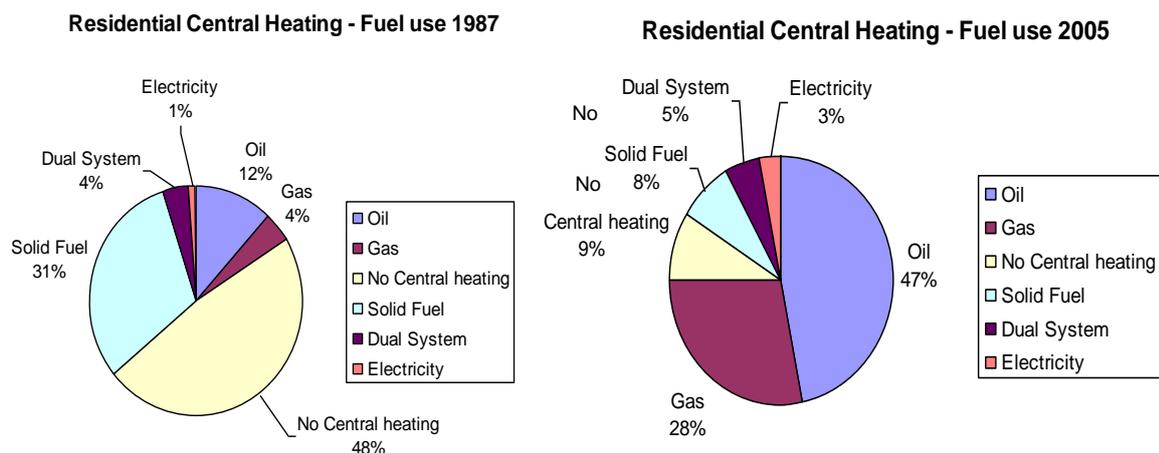


Figure 2.1. Residential Central heating in Ireland, 1987 Vs 2005. Source [14]

There has also been a clear shift away from solid fuel heating systems, based on peat and coal, towards oil- and gas-fired systems. [Figure 2.2](#) highlights the significant rise in oil and gas usage and corresponding decline in peat and coal consumption. There has been

very low usage of biomass in this sector over the period (<1% of total residential fuel consumption) but the SEAI predicts that this will grow into the future due to increasing use of biomass for residential heating.

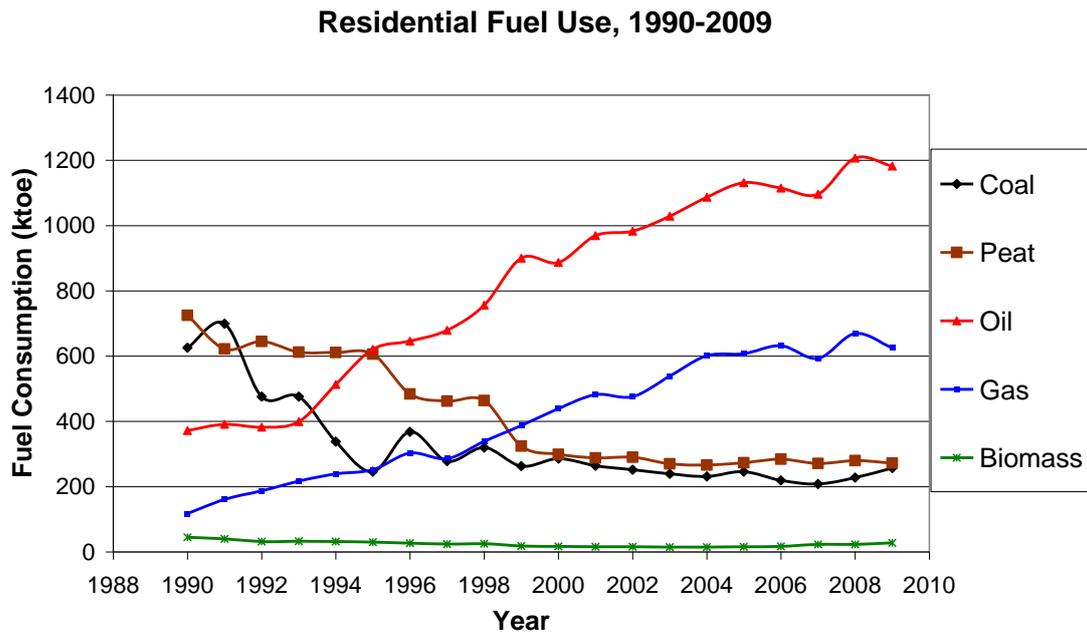


Figure 2.2. Residential fuel use by fuel type. Source: [15]

An industry sales report compiled for Glen Dimplex [16] paints a similar picture. The report does not include data for sales of domestic biomass boilers, as at the time the report was compiled, biomass boilers were very rare in the Irish market. It is interesting to note that sales of gas boilers were increasing more rapidly than sales of oil

boilers. This can be attributed to the continuing expansion of the Bord Gáis transmission network [17], as well as the higher efficiencies gas boilers offered. Solid fuel boiler sales had declined and sales had levelled off at approximately 4,000 units per year.

Trends in Irish Boiler sales 1991 - 2004

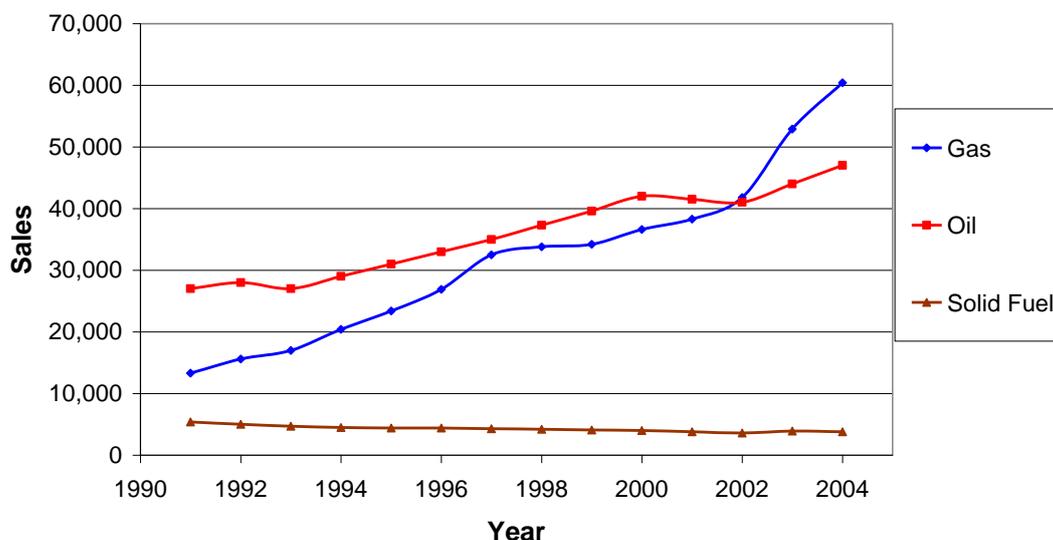
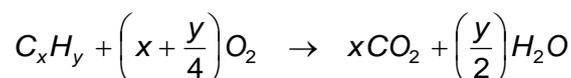


Figure 2.3. Trends in Irish central heating boiler sales 1991-2004. Source [16]

It is clear, therefore, that the contribution of the heating sector, both residential and commercial, has a very significant role to play in Ireland's total energy use. Ireland's strong dependence on imported fossil fuels as a source of energy mean that oil and gas account for a particularly large proportion of energy usage. As the residential and commercial heating sectors account for almost 25% of Irish energy usage, and non road transport just 5%, the main focus of this report will be on deriving new emission factors for combustion appliances in Ireland.

2.3 Links between combustion and emissions

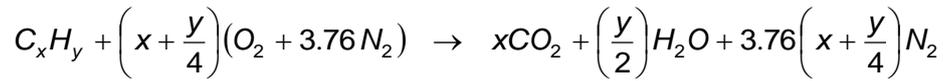
Combustion can be defined as 'the rapid union of a substance with oxygen accompanied by the evolution of light and heat' [18]. In other words, a hydrocarbon fuel is reacted with oxygen to produce heat, and chemical compounds known as products of combustion. The stoichiometric amount of oxidiser is that amount needed to completely burn a quantity of fuel [19]. Combustion of a hydrocarbon fuel produces carbon dioxide (CO₂), water vapour (H₂O) and heat. For a hydrocarbon fuel with general chemical formula C_xH_y, the ideal stoichiometric combustion equation can be expressed as:



Equation 1: Ideal combustion equation with oxygen

For most combustion appliances, the oxygen source is from the air in the atmosphere, which is comprised of approximately 21% oxygen and 79% nitrogen, so that the ratio of nitrogen to oxygen is 3.76 (79/21). During

the combustion process, the nitrogen behaves as an inert gas and, in general, does not react with other elements [20]. Hence, the combustion equation becomes:



Equation 2: Ideal combustion equation with air

While an ideal combustion process occurs with the stoichiometric quantities of fuel and air, it is impractical to achieve this in a real combustion system [21]. If more air than the stoichiometric quantity is supplied to the reaction, the mixture is said to be lean, while supplying less than the stoichiometric

amount of air results in rich combustion. A useful parameter for describing combustion conditions is the air:fuel ratio (AFR), which is defined as the ratio of the air mass flow rate to fuel mass flow rate [22]. It is given by Equation 3.

$$AFR = \frac{\dot{m}_a}{\dot{m}_f}$$

Equation 3: Air Fuel Ratio

The AFR depends on the chemical composition of the fuel being combusted but for liquid fuels, such as kerosene and diesel, that are used for domestic heating, the AFR typically lies between 14 and 15. As different fuels have different AFRs ($AFR_{\text{natural gas}} \approx 17$, $AFR_{\text{wood}} \approx 6$), based on the carbon and

hydrogen content of that fuel, a new parameter is needed to compare AFR values for different fuels. This parameter is called the relative air:fuel ratio λ , and is the ratio between the actual AFR and the stoichiometric AFR. It is given by Equation 4.

$$\lambda = \frac{AFR_{\text{actual}}}{AFR_{\text{stoichiometric}}}$$

Equation 4: Relative air:fuel ratio λ

Using this parameter effectively normalises the AFR, regardless of the fuel type in question. For a lean combustion process, the relative AFR is greater than one ($\lambda > 1$), while for a rich mixture, it will be less than one

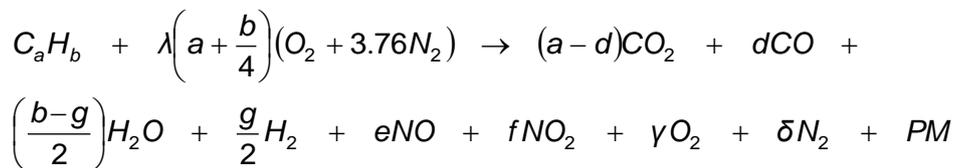
($\lambda < 1$). The fuel:air ratio (FAR) is simply the inverse of the air:fuel ratio, as previously defined. Similar to the relative air:fuel ratio λ , the equivalence ratio ϕ can be defined as:

$$\phi = \frac{FAR_{actual}}{FAR_{stoichiometric}}$$

Equation 5: Equivalence ratio, ϕ

For complete combustion to occur, as per Equation 2, the temperature, time and oxygen supply must all be sufficient, a situation which does not occur in practice [23]. A combustion process is complete when all the carbon in the fuel is oxidised to CO₂, all the hydrogen to H₂O and the nitrogen remains unchanged [20]. In a real combustion system, however, other unwanted products of combustion called ‘air

pollutants’ are formed as well, such as carbon monoxide(CO), nitric oxide(NO), nitrogen dioxide (NO₂), hydrogen (H₂), unburnt hydrocarbons (UHC) and particulate matter (PM) may be formed. A pollutant may be defined as ‘any abnormal material or property that reduces the usefulness of the air resource’ [23]. The general combustion equation then becomes:



Equation 6: General combustion equation for real combustion process

2.4 Combustion technologies

The basic operating principle of a heating boiler is that fuel is burnt in a combustion chamber which is surrounded by a water jacket. The heated water is then pumped around a circuit in a building, passing through heat emitters, such as radiators, and thereby heating rooms. The hot water also passes through a loop in a hot water

cylinder, producing hot water for domestic use, such as washing, etc. In general, boilers are described by the fuel they burn, either oil, gas or solid fuel.

The achievement of clean and efficient burning in an oil-fired boiler depends mainly on atomisation of the fuel, which involves intimate mixing of the carbon in the fuel with oxygen in the air supply. Early oil-fired

boilers used vapourisation to achieve this. On start-up, the oil was preheated electrically to produce a vapour. This vapour was then ignited and once combustion had been initiated, the heat generated by the flame allowed further vapourisation to continue. However, modern boilers use pressure jet atomisation, where the oil is supplied to a fine nozzle under pressure. As the oil is accelerated through the nozzle, its

pressure drops and this causes the oil to atomise into a fine mist. The nozzle is designed to give a swirling motion to the spray of droplets produced. The air supply is provided in a contra-rotating swirl. The resulting turbulent motion provides good mixing conditions between the atomised fuel and the air supply. A schematic diagram of a pressure jet atomising oil burner is shown in [Figure 2.4](#).

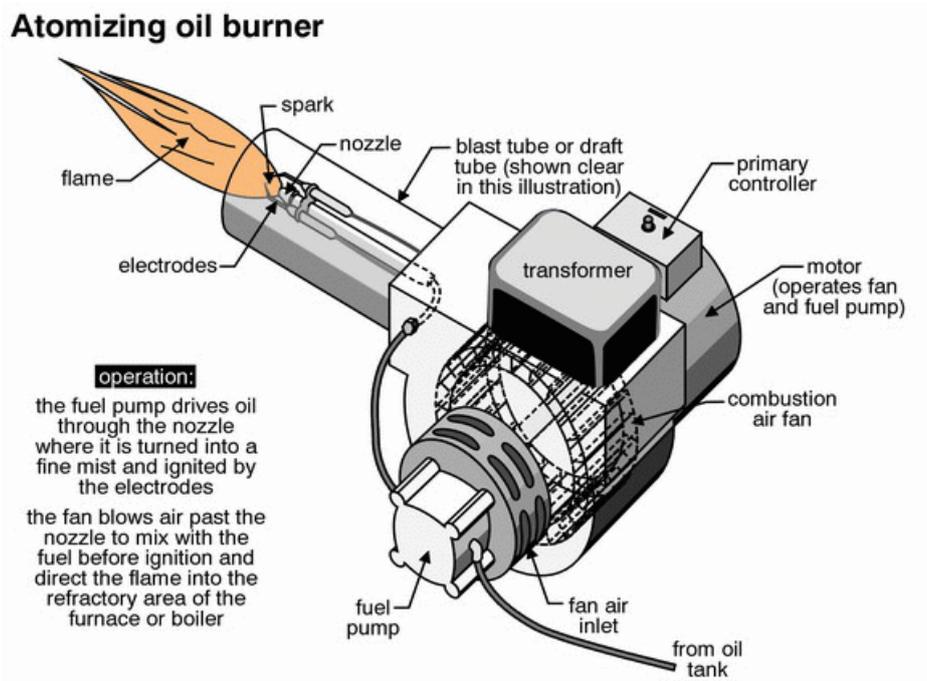


Figure 2.4. Atomising oil burner. Source: [24]

A gas boiler burns either natural gas or bottled propane, usually in an atmospheric burner. In this burner, gas is supplied to a nozzle which is fitted to a manifold. It flows through a venturi where a supply of primary air is induced. At this stage, the mixture is fuel-rich, typically about 50% of the stoichiometric air requirement. This level of primary aeration results in short flames and

prevents soot formation, resulting in the familiar blue flame [25]. The fuel air mixture is then delivered by the manifold to a series of outlet ports, where the flame produced entrains a secondary supply of air from the surrounding space. Combustion is initiated by a small pilot flame. A schematic diagram of this type of burner is shown in [Figure 2.5](#).

Gas burners

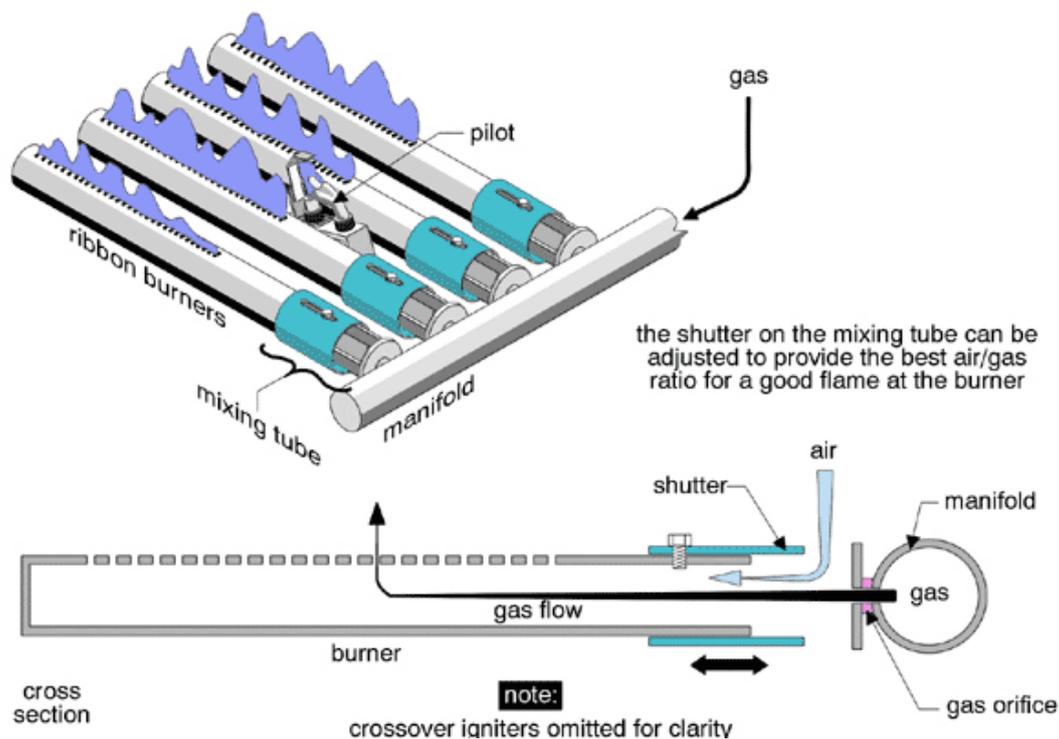


Figure 2.5. Atmospheric gas burner. Source: [24]

Solid fuel systems mainly consist of grate firing, although fluidised bed combustion is sometimes used on larger, industrial-scale units. There is a large range of solid fuel-fired boiler technologies, ranging from an open fireplace with a back boiler up to a modern, fully-automated pellet boiler. Traditionally in Ireland, all solid fuel boilers were either peat- or coal-fired and the fuel was typically fed in manually. These boilers are labour intensive to use and have low efficiencies, typically 60% or less [26]. Recently, however, a new type of solid fuel boiler has arrived in the Irish market, the wood-pellet boiler.

Wood-waste materials, such as sawdust, which has a low energy density, can be compacted into high energy content pellets which facilitate storage, transport and combustion [27]. In the pelletising process,

the raw material is dried to a moisture content of 9-12% and milled to produce particle sizes of typically less than 4mm. These particles are then screened to remove oversized particles and extruded through a die. The friction generated between the wood material and the die walls results in a rise in temperature which causes the natural lignin in the wood to soften and bond the particles together [28]. Corn or maize starch can also be used as a binding agent for other pellet fuels.

The pellets are of uniform size and density, with a diameter of 6mm and a length of not more than 30mm. Wood-pellets sold in the EU must conform to the CEN/TS 14961 standard and [29] must have a moisture content of less than 10%. Their lower calorific value depends on the pellet

moisture content and the raw material used but is typically around 18 MJ/kg.

In order to ensure complete combustion of the pellets with low emissions, the supply of combustion air is divided into two stages. The combustion chamber has a primary and secondary zone, each with its own air supply. In the primary zone, the remaining moisture in the pellet is evaporated and the dry pellets are then decomposed into combustible, volatile components and char. In the secondary zone, the flammable gases are burnt with excess air. To get optimum combustion, the secondary air nozzles are designed so that there is proper mixing of the air with the flue gases. The amount of excess air in the secondary zone is also important because, if it is too low, then there will be increased emissions of carbon monoxide (CO) and unburnt hydrocarbons (UHCs). However, if excess air levels are too high, then the NO_x emissions will increase [30].

The pellets are typically fed from a storage hopper via a rotating auger to the boiler, where they drop onto the combustion bed. Such a system is called a top feed system but bottom feed systems are also available, where the pellets are fed from below the combustion bed. The advantage of this system is that the fuel supply is very consistent and that the new pellets push the ash off the combustion bed and into an ash pan below. The major disadvantage though is that burn back is significantly more likely and, hence, top feed systems are generally preferred on safety grounds. Wood-pellet boilers are typically controlled by temperature settings and most pellet burners have two or more discrete power levels. For each of these power levels, the settings for

the fuel and air supply remain fixed to give different heat outputs.

Wood-pellet boilers also require more maintenance than oil- or gas-fired systems. Depending on the ash content of the fuel, the ash pan has to be emptied at regular intervals, and the heat exchangers require more frequent servicing and cleaning due to the higher concentration of particulate matter and soot being released from the combustion chamber. In more advanced boilers, these functions are carried out automatically, while in simpler, less advanced boilers, the user must clean it out manually.

In the past, heating boilers tended to have very high flue gas temperatures, typically greater than 150°C at the flue exit. At these temperatures, water vapour in the flue gas remains as a vapour and the latent heat cannot be reclaimed [31]. This was mainly done due to impurities in the fuel (e.g. sulphur), which condense and form contaminants at low temperature, which lead to corrosion and fouling of the heat exchanger. To overcome this, additional corrosion prevention measures need to be incorporated into the design of the boiler, increasing costs. As of 31st March 2008, all oil and gas boilers for sale in the Republic of Ireland must have a minimum seasonal efficiency of 86% to comply with the relevant building regulations [32]. This, in effect, means that oil and gas boilers available on the market now will all be of the condensing type.

Condensing boilers offer significant increases to the efficiency of a heating boiler. In addition to increasing the available energy available for extraction from

combustion of a fuel, they can also reduce pollutant emission levels by partially or totally dissolving them in the condensed water, as reported by Che *et al.* [31]. In addition, recent studies have examined whether or not operating a biomass boiler in condensing mode could actually reduce the level of particulate matter released to the atmosphere. Grohn *et al.* [33] hypothesised that when water vapour condenses from the flue gas, not only does it improve the heat recovery and boiler efficiency but it also reduces the heat exchanger slagging and fouling by generating a film of water on the heat exchanger surface that carries away deposited particles. They found that particle number concentration could be reduced by up to 40% through appropriate use of the condensing heat exchanger. Research in the Netherlands [34] looked at reducing particulate emissions both by condensation onto heat exchanger walls and by water vapour condensing onto particles already formed in the exhaust gas stream. A 180kW biomass boiler was tested and it was found that fine particulate emissions could be reduced by 65%. However, in order to achieve this, it was necessary to use a heat exchanger with very small passageways (2.22mm), as this gave a large surface area for particles to condense onto. Typically, biomass heat exchanger tubes for that size of boiler are in the order of 40mm in diameter. However, it remained unclear whether long-term build up of particles, which could clog the very fine passageways in the heat exchanger, outweighed any short-term benefit.

2.5 Pollutant formation

The ideal combustion conditions described in [Section 2.3](#) are never achieved in practice and, as a result of impurities in the fuel, poor AFR control and incomplete combustion, pollutants, such as NO_x and PM, are formed. Insufficient oxygen is one of the main reasons for incomplete combustion. If the combustion mixture is rich (i.e., $\lambda < 1$), then not all of the carbon in the fuel will be oxidised to CO₂. Rather, it ends up as carbon monoxide (CO) or as carbon particles (soot) in the combustion products. To minimise CO formation, lean mixtures ($\lambda > 1$) are used. This also reduces the amount of unburnt fuel in the exhaust gas. Changes in the AFR can have a significant impact on emission levels.

[Figure 2.6](#) shows the variation of three common pollutants, CO, HC and NO_x as the amount of air available for combustion changes. When the mixture is rich (fuel/air equivalence ratio > 1), there is incomplete combustion and the levels of unburnt hydrocarbons and carbon monoxide begin to rise rapidly. In this region, NO_x levels are low. However, when a slightly lean mixture is used, carbon monoxide levels fall away to zero, yet NO_x reaches a peak at approximately 5% excess air before reducing again at higher AFRs. However, there is an upper limit to the AFR, after which flame extinguishing can become a problem, so choosing a suitable AFR involves a trade-off between NO_x and other pollutants.

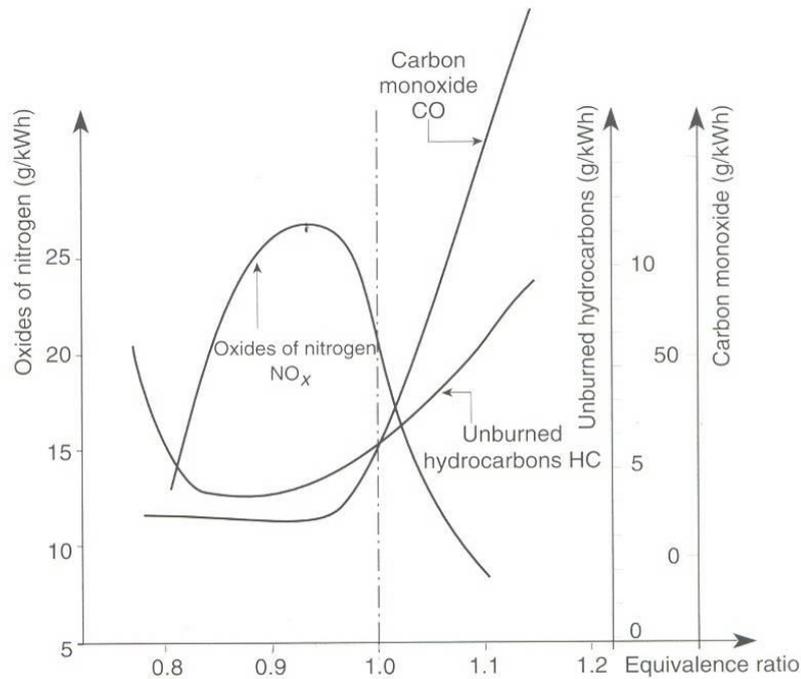


Figure 2.6: Variation of HC, CO and NO concentration plotted against fuel/air equivalence ratio. Source: [22]

The levels of carbon monoxide (CO) in the flue gas can also give a good indication as to the relative amount of particulate matter. PM arises mainly from incomplete combustion, and [Figure 2.7](#) shows the trends in carbon monoxide emissions from the combustion of wood as a function of the AFR. Similar to [Figure 2.6](#), the levels of carbon monoxide rise sharply as the mixture begins to richen. In the case of the wood burnt in a study by

Nussbaume [35], the CO minimum occurred when there was between 50% and 100% excess air. For other combustion systems, the minimum might be closer to the stoichiometric mixture due to better mixing of the fuel and air prior to combustion. This is particularly true for oil and gas combustion, as these mix with the air better than solid fuels.

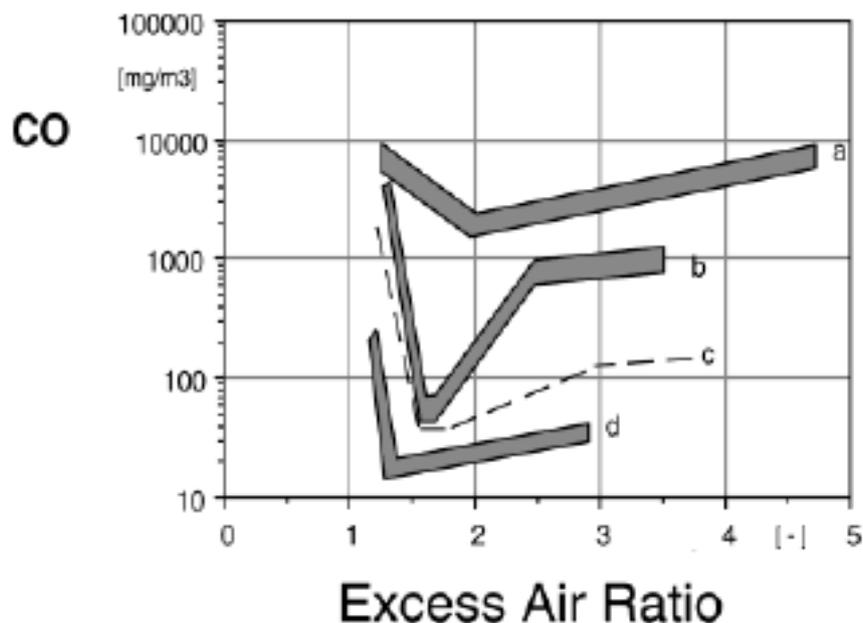


Figure 2.7. Carbon monoxide emissions as a function of excess air ratio for different furnace types (a) Wood stove (b) Downdraft boiler (c) Automatic wood furnace (d) Advanced automatic wood furnace. Source: [35]

Because the rate of pollutant formation is heavily dependent on the AFR, it is very important that a combustion appliance is set up correctly. During transient operation of a boiler, e.g. during start-ups, the AFR value can be wrong temporarily, which often leads to higher levels of emissions during those operating modes. If the AFR in a boiler (regardless of fuel type) is not set correctly, then the amount of pollutants emitted will increase. For example, in an oil burner, the flow rate of fuel into the combustion chamber is fixed by the pump pressure and the nozzle size. The only variable is the air supply, which is adjusted by changing the position of an air damper. Over time, these burner components can wear and the AFR will move away from its correct setting. Therefore, it is important during boiler servicing that the flue gas composition is checked and air supply adjusted where necessary, to ensure that the boiler is

operating at optimal efficiency while emitting minimal pollutants. High boiler efficiency is of great importance, not only for reduced heating costs, but also in terms of the emissions produced, as with a higher efficiency unit less fuel is burnt to meet the heating requirements of a building.

2.6 Human health impacts of air pollution

This section begins with a brief outline of the human respiratory system. Sections [2.6.2](#) and [2.6.3](#) that follow describe the effects that oxides of nitrogen (NO_x) and particulate matter have on the human body.

2.6.1 The human respiratory system

The hazard caused by inhaled pollutants depends on both their chemical composition and on the site at which they deposit in the respiratory system [36]. The main

constituents of the respiratory system are the lungs and the airways that supply the lungs. Its principal function is to transport oxygen from the atmosphere to the bloodstream and simultaneously release carbon dioxide from the bloodstream to the atmosphere. It can be divided into three regions:

1. Head airways region
2. Lung airways or tracheobronchial region
3. Pulmonary or alveolar region.

The first region includes the nose, mouth, larynx and pharynx. Inhaled air is warmed

and humidified in this region. The second region comprises the airways from the trachea to the terminal bronchioles, and resembles an inverted tree with a single trunk (trachea) subdividing into smaller and smaller branches (bronchioles). The final region is the pulmonary region and it is here that gas exchange processes (oxygen/carbon dioxide) take place [36]. A schematic diagram of the respiratory system is shown in [Figure 2.8](#).

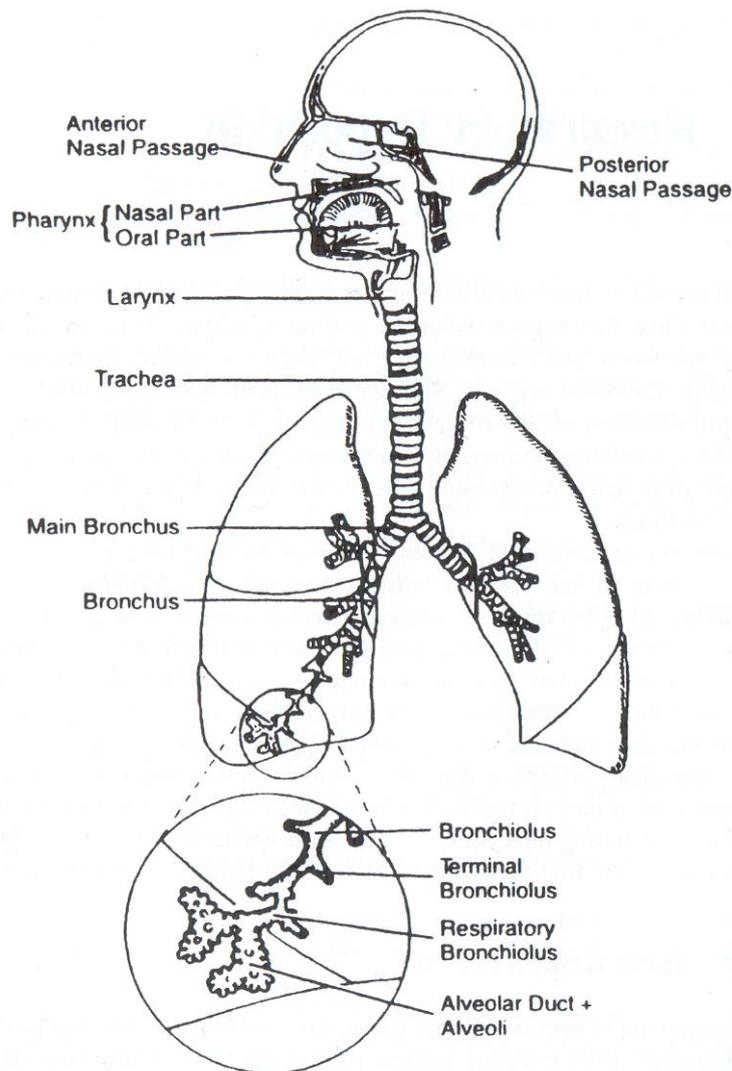


Figure 2.8. Outline of the human respiratory system. Source:[36]

A person inhales between 6m³ and 12m³ of ambient air per day, depending on age and physical activity [37]. The surface area of the pulmonary region that is used for gas exchange is approximately 75m². Inhaled air follows a flow path that goes through a sequence of 23 airway branchings from the trachea to the alveolar surfaces. The first 16 of these take place in the tracheobronchial region, with the remainder occurring in the alveolar region [36]. Particle behaviour in the lungs is dependent upon the aerodynamic characteristics of the inhaled particles, while the main factor for gaseous pollutants is their solubility in the linings of the different regions of the respiratory system. Highly soluble gases, such as sulphur dioxide (SO₂), are absorbed in the upper airways, while insoluble gases, such as NO₂, penetrate into the pulmonary region. The dissolved gases may be eliminated by biochemical processes or may diffuse to the circulatory system [18].

In terms of particle emissions, the body has a number of defence systems whose purpose is to remove particles as quickly as possible. Deposition in the head airways region is highly variable and depends on a number of variables, including breathing rate and particle size. Particles are warmed and humidified on entry to the nasal passage, and hygroscopic growth begins for soluble particles. Very large particles may be removed by impaction with nasal hairs, while smaller particles move further into the body. The retention time for particles in the lungs depends on a number of factors, such as their physicochemical properties, their location and the type of removal mechanism needed. The airway surfaces in the first two regions are covered in a layer of mucus that removes larger particles from the airways by inertial impaction. This mucus layer slowly

moves upwards, transporting particles towards the pharynx where they may be coughed up or swallowed [36].

The alveolar region lacks this mucus layer, due to its gas exchange function. In this region, mobile macrophage cells take up particles by phagocytosis and remove them from the alveoli to the mucus layer by active transport. Macrophages are white blood cells, approximately 21µm in diameter [38], whose role is to engulf and digest (phagocytose) particles which have deposited in the lungs. Soluble particles are removed by dissolution and transport to the lymphatic drainage system. However, fine, insoluble, solid particles can deposit deep in the alveolar region of the lungs and it can take months or years for the body to remove them. The process of clearing these particles can induce secondary physiological responses including coughing and inflammation [37]. More detail about the health implications of these ultrafine particles is provided in [Section 2.6.3](#).

The five deposition mechanisms by which particles may deposit on a surface are:

1. Inertial impaction
2. Gravitational settling
3. Diffusion
4. Interception
5. Electrostatic attraction.

Interception and electrostatic attraction are of minor interest when considering deposition of particles within the human body, the majority of particles being deposited by either inertial impaction, diffusion or gravitational settling. When air is inhaled through the nose or mouth, it must pass through a branching airway system before it reaches the lungs. Larger particles will often travel for a short distance before

changing direction, as a result of their own inertia, so particles that are near the airway surfaces deposit by inertial impaction. This mechanism removes larger particles that are moving close to airway surfaces. The effectiveness of this mechanism depends on the ratio of the particle stopping distance to airway dimensions and is highest in the bronchial region [36].

In the smaller airways found after the trachea, gravitational settling becomes a more important deposition mechanism. Here, flow velocities and airway dimensions are small, and for horizontally inclined airway passages, in particular, the particles fall from the air stream under the influence of gravity and impact on the cell walls. This mechanism becomes very important at regions far from the trachea. Hygroscopic particles will grow in the saturated environment found in the trachea and lungs and will deposit both by impaction and settling. Finally, for particles less than 0.5µm in diameter, the dominant deposition mechanism is by diffusion and is governed by the particle's geometric, rather than aerodynamic, diameter. They are most likely to deposit on the smaller airway walls, where distances are short and residence times relatively long [36].

2.6.2 Health impacts of NO_x

Of all the nitrogen oxides only NO and NO₂ are thought to be harmful to human health [39]. NO is formed in combustion processes and subsequently oxidised to NO₂ in the atmosphere. In-vitro studies have shown that while NO₂ induced chromosome aberrations and DNA damage, no such effect was found with NO, while in-vivo studies showed that the mutagenic effect of NO was insignificant

compared to NO₂ [40]. Hence, the following discussion focuses on the health impact of nitrogen dioxide. NO₂ is a major indoor air pollutant and is generated from cigarette smoking as well as natural gas and kerosene burning, which are frequently used as energy sources for cooking and heating. It is a strong respiratory irritant which can reach deep into the human respiratory tract and be retained in the bronchotracheal and alveolar regions [41]. Animal tests have found that short-term exposure to concentrations above 200 mg/m³ is lethal to rats [42], while sub-lethal, short-term exposure can cause reversible effects, such as altered lung function, airway hyper-responsiveness, epithelial damage and inflammation [41]. Lung oedema, or accumulation of fluid in the lungs has been observed after short-term exposure to concentrations of 20 mg/m³ and above [42].

NO₂ is almost completely absorbed in the respiratory tract [40]. Up to 40% may be absorbed in the nasal passage, while the remainder deposits in the terminal bronchioles and alveolar region [41]. It has also been found that NO₂ effects tend to occur many hours after exposures have ceased, so that persons exposed to NO₂ are often unaware of the severity of their exposure for some time after [39]. Gauderman *et al.* [43] found that there was a 1% reduction in lung function in children aged between 10 and 18 years for every 20 µg/m³ increase in NO₂ levels. The current WHO guideline values for NO₂ are a one-hour level of 200 µg/m³ and an annual average of 40 µg/m³ [44].

A study in Oslo, Norway [45] analysed the effect on nitrogen dioxide (NO₂) on cause specific mortality in a population-wide

sample based on a short-term causal model. They examined a population of 143,842 people in 470 areas over a six-year period and found a strong link between atmospheric concentrations of NO₂ and causes of death. Their results found that for people under 70 years of age there was no adverse health impacts when average ambient concentration remained below 40 µg/m³, while for people over 70 years there was a linear increase in pollution-related illness once ambient concentrations exceeded 20 µg/m³. Other European studies have shown that mortality rates increased by 2.0-7.6% for a 100 µg/m³ increase in atmospheric NO₂ levels [46, 47].

2.6.3 Health impacts of PM

Particle pollution has been associated with the deterioration of air quality for centuries. In 1661, John Evelyn published the first study on air pollution [48] and described the air pollution in London by saying: "London fires, there results a great quantity of volatile Salts, which being sharp and dissipated by the Smoake doth infect the Aer, and so incorporate with it, that though the very Bodies of those corrosive particles escape our perception, yet we soon find their effects, by the destruction of all things they do but touch; with their fuliginous qualities" [37]. In more recent times, the Great London Smog of 1952 killed over 4000 people in just four days as a result of excessive coal combustion [49]. The ambient particulate concentration at the time was estimated to have exceeded 1600µg/m³, or 80 times higher than the current recommended guideline value [50, 51]. These events focussed attention onto the subject of air pollution and quality, and resulted in an

increased scientific interest on research into air pollution and its associated health risks.

After many years of research into the health effects of particulate matter, a statistically significant correlation between levels of fine particles and adverse health effects, such as increased morbidity, reduced lung function and mortality from pulmonary and cardiovascular diseases has clearly been established [43, 52-58]. Initial research into the health impact of particulate matter used epidemiology methods to investigate the quantitative factors that control the frequency and distribution of disease, and derived correlations between ambient air quality and hospital admissions/mortality rates over an extended period of time [52, 59-61]. In the seminal paper (Harvard Six Cities study), Dockery *et al.* [52] examined the link between ambient air pollution and mortality for over 8,000 subjects across six US cities for a period of 14-16 years. They found that mortality was most strongly linked to cigarette smoking, but that once the effects of cigarette smoking were filtered from the sample ambient fine particulate matter, concentration was very strongly correlated with mortality. In essence, the study found that an increase in ambient particulate concentration of 10µg/m³ increased overall mortality rates by approximately 1.5%, with the following mortality rate increases for particular ailments: mortality from pneumonia by 4%, from chronic obstructive pulmonary disease by 3% and from ischemic heart disease by 2% [62]. Figure 2.9 shows the very linear correlation between mortality rate ratio and ambient concentration of fine particles. Similar results were observed in a much larger study, involving 552,138 people across 151 US metropolitan areas, carried out by Arden Pope *et al.* [53]. They found

that a $10\mu\text{g}/\text{m}^3$ increase in ambient fine particulate air pollution resulted in a 6%

increase in cardiopulmonary mortality and an 8% increase in lung cancer mortality [56].

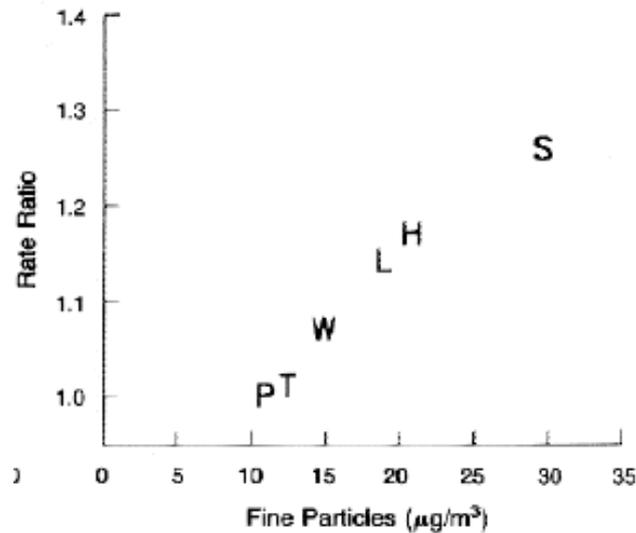


Figure 2.9. Estimated adjusted mortality-rate ratios and pollution levels in the Six Cities used in the Harvard study. The cities are as follows: P, Portage, Wisconsin; T, Topeka, Kansas; W, Watertown, Massachusetts; L, St. Louis, Missouri; H, Harriman, Tennessee; S, Steubenville, Ohio. Source: [52]

In Ireland, a shift from liquid fuels to bituminous coal for residential water and space heating in Dublin in the 1980s resulted in a deterioration in air quality [63]. In response to this, the Irish Government banned the marketing, sale and distribution of bituminous coals within the city of Dublin on September 1st 1990 [64]. Clancy *et al.* [65] examined the air quality related death rates before and after this legislation was enacted and found similar trends to the American research. They found that average black smoke concentrations in the city were reduced by $35.6\mu\text{g}/\text{m}^3$ (70%) as a result of the ban, and found that respiratory and cardiovascular related deaths decreased by 15.5% and 10.3% respectively, which corresponded to approximately 116 fewer respiratory deaths and 243 fewer cardiovascular deaths per year in Dublin.

Epidemiology uses observational studies of real-world exposures among free-living subjects, [62] but it cannot prove causality [37]. Initially, it was thought that sulphur dioxide was the most toxic pollutant and that particulate acted merely as a relatively benign agent for gaseous sulphur dioxide to condense onto and be transported deep into the lungs, whereas in the absence of particle carriers, the sulphur dioxide would be extracted in the upper respiratory tract [51]. However, more recent research has found that inhalation of fine particulate matter induces both pulmonary and systemic inflammation and oxidative stress, leading to direct vascular injury, arteriosclerosis and autonomic dysfunction [60, 66].

To determine the exact effect exposure to particulate matter has on the human body, toxicological studies need to be carried out. These can be categorised as either *in vivo* or

in vitro studies. *In vivo* experiments investigate effects in living organisms, such as experimental animals, while *in vitro* experiments are conducted in organs, tissues, cells, etc., that are isolated from the living organism [67]. The difficulty with toxicological studies is the inability to generate fine particulate matter with the same physical, chemical and health properties as the fine particles found in ambient air [62].

In general, *in vivo* studies have found that exposure to particulate matter result in oxidative stress and inflammation of cells in the respiratory system [68-74]. When cells come into contact with fine particles, reactive oxygen species (ROSs) form within the lung epithelial cells that try to de-activate and eliminate the invading foreign material [75]. These ROSs contain strong oxidants, such as the OH radical, which can cause breaks in DNA strands and can lead to the production of signalling molecules that give rise to inflammation [58]. The inflammatory response leads to damage in the epithelial cell layer, which weakens the defences of those cells. Hence, particle deposition on the respiratory epithelium can trigger a cascade of events in many different cells, which can bring changes to tissues and organs further away from the initial stimulus [67]. Ghio *et al.* [76] found that exposure to PM changes the characteristics of blood by increasing fibrinogen levels. Fibrinogen is a key component in blood coagulation and platelet thrombosis, and a major determinant of blood viscosity. Blood viscosity has been associated with cardiovascular disease and formation of atherosclerosis [66, 77].

In vitro studies generally expose macrophages and respiratory epithelial cells

to fine particles and observe the response of the cell. Macrophages are the main cell type for clearing large particles in the lungs, while epithelial cells absorb fine and ultrafine particles [78]. Similar to *in vivo* experiments, *in vitro* studies have shown that exposure of cells to fine particles causes an inflammatory response [78-82]. However, *in vitro* experiments have a number of limitations. For example, the particle exposure does not mimic the exact conditions found in the body and, in addition, they also lack the cellular interactions and neurological signals that are of importance in animals and humans. The exposure of pulmonary cells to fine particles during *in vitro* studies is not necessarily the same as their exposure while in the body, due to deposition mechanisms found in the upper respiratory tract.

The adverse health effects caused by inhaled particles depend on the deposition and retention of particles in the lung, which themselves depends on their size, shape, density and hygroscopicity. Nanoparticles can catalyse chemical reactions on their surface, and as these smaller particles have a large specific surface area (surface area per unit mass), they are more reactive than the same mass of material made up of larger particles. Hence, as both *in vitro* and *in vivo* studies have found, they induce a more pronounced inflammatory response than larger particles of the same material [54, 67, 83-86]. There is still some debate as to what is the best metric to assess the health impact of particulate matter [58]. Traditionally, toxicology studies have focussed on the mass concentration of particles, but it has been suggested in the literature that a metric based on the surface area of particles would yield better information [54, 59].

The atmosphere that particles are exposed to in the human respiratory system is very moist, and in the lower respiratory tract and lungs the equilibrium relative humidity (RH) is approximately 99.5% [87]. Hygroscopic particles, such as concentrated sulphuric acid droplets, calcium sulphate or sodium chloride, tend to grow after inhalation by the addition of water, by virtue of this high humidity in the respiratory tract and, thus, are deposited higher in the tract than their diameter prior to inhalation would suggest. For example, Dautrebanda and Walkenhorst [88] compared the deposition of coal dust and dry sodium chloride particles of the same size and found it necessary to correct by a factor of seven to account for the growth of salt particles in the respiratory tract [39].

More recently, Londaal *et al.* [89] examined the hygroscopic behaviour of particulate matter from biomass combustion and how it influenced its deposition characteristics. Emissions from two combustion conditions were examined and respiratory deposition measurements carried out on 10 healthy subjects. They found that the particles produced during good combustion conditions were composed almost entirely of soluble alkali salts (typically KCl and K_2SO_4), which grew by up to a factor of 5 when exposed to the 99.5% relative humidity found in the lungs. Due to their high solubility, these

particles quickly dissolved and were removed by circulation. Conversely, particles produced by poor combustion conditions mostly consisted of carbonaceous organic compounds and soot which exhibited a lower growth potential, and didn't grow at all once particles reached 400nm in diameter. As a result of their low solubility, they could be expected to remain in the lungs for long periods of time, especially if deposited in the non-ciliated alveolar region [90].

2.7 Environmental impacts of air pollutants

2.7.1 Environmental effects of NO_x

In addition to being harmful to health, NO_x also has a detrimental effect on the environment. It can react with volatile organic compounds (VOCs) in the atmosphere to form ground level ozone or smog. The word 'smog' comes from 'smoke' and 'fog' and reduces visibility by blocking the transmission of light. In addition, green vegetation, such as tree foliage, can absorb ozone onto their leaves and impede growth. Depending on the concentration of NO_x (particularly NO_2) that the plant is exposed to, the damage can range from reduced growth rates to death of plant cells. As [Figure 2.10](#) shows, if exposure time is long enough, damage can occur even when the NO_2 concentration is very low.

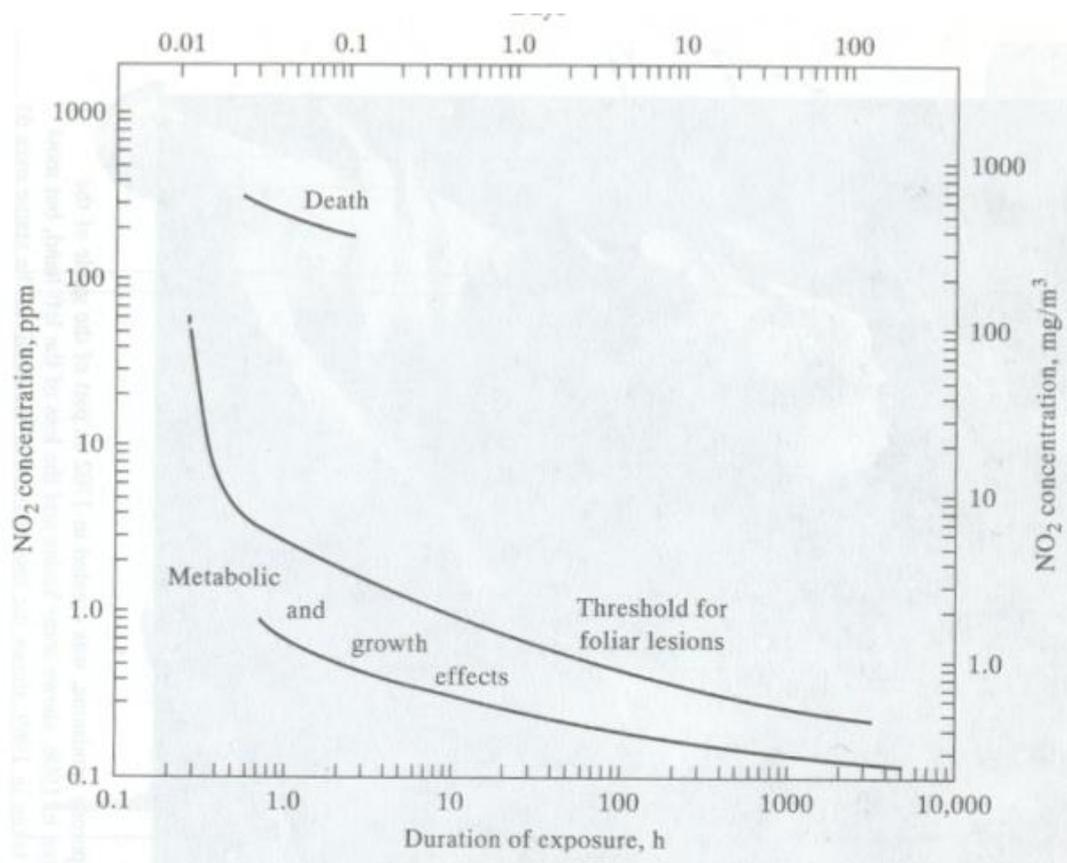
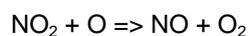
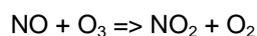


Figure 2.10. Threshold curves for the death of plants, foliar lesions and metabolic or growth effects, as related to the nitrogen dioxide concentration and the duration of the exposure. Source: [91]

NO_x also has a damaging effect on the ozone layer in the upper atmosphere. The purpose of the ozone layer is to absorb the damaging solar UV radiation, but pollutants, such as NO_x and chlorofluorocarbons (CFCs), emitted to the atmosphere as a result of human activity have been depleting this layer and allowing harmful radiation to reach the earth's surface. Emission of NO_x into the upper atmosphere by aircraft can destroy the ozone layer by the following reaction:



In the first step the ozone is destroyed, while in the second step NO is regenerated which

causes further ozone layer damage [19]. Hence, a positive feedback cycle gets established, which results in significant damage to the ozone layer.

NO_x also contributes to acid rain and, as outlined in Section 2.1, one of the main reasons why emissions monitoring became so widespread was due to acid rain and the acidification of Scandinavian lakes. Acid rain is formed when NO_x reacts with sulphur dioxide and can erode buildings as well as affecting lakes and soils. NO_x pollution can also lead to reduced fish populations due to the fact that eutrophication in water bodies is accelerated by nitrogen. Eutrophication is 'the enrichment from anthropogenic sources which leads to the excessive growth of algae

at the water surface' [92]. Finally, NO₂ is also a greenhouse gas that, along with CO₂ and other greenhouse gases, is causing a rise in global temperature and associated climate change problems.

2.7.2 Environmental effects of PM

Particulate matter emitted to the atmosphere has an adverse effect on the environment. It contributes to smog and reduced visibility, as well as damaging vegetation and ecosystems. Higher atmospheric concentrations of particles reduced visibility by scattering light before it reaches the eye. For example, in foggy weather, a car approaching with its head lights on is not visible at first because all of the light from the

headlights has either been scattered or absorbed by the water particles that make up fog [91]. Similarly, increased levels of particulate matter in the atmosphere have a similar effect. For very high particle concentrations, such as that found during the London Smog of 1952 [49], visibility is greatly reduced. Thankfully, such levels of ambient particulate matter are generally not encountered any more, at least in the developed world, but even in cities such as Los Angeles, visibility can be greatly reduced by elevated levels of particles in the atmosphere. [Figure 2.11](#) shows an example of this reduced visibility. Atmospheric smog can also be produced by chemical reactions among primary pollutants, both gaseous and PM, to form secondary particles [91].



Figure 2.11. Limited visibility in Los Angeles caused by high particulate matter concentration. Source: [93]

Particulate matter also has significant adverse effects on plants and vegetation. Fine particles emitted from combustion sources have a particular chemical composition that is directly related to the composition of the fuel that was burnt. In the atmosphere, however, these particles grow, as well as new particles forming, from reactions with oxides of nitrogen and sulphur. Coarse PM can be considered a primary particle, as it is produced and emitted from a source as a fully formed particle, and it is these coarse particles that dominate the mass fraction of atmospheric PM. Fine PM, on the other hand, is more diverse and can be considered secondary in nature. It contains many different constituent parts, having gone through gas-to-particle conversion and subsequent growth by condensation and coagulation, and can contain products of incomplete combustion, inorganic salts, volatile organic compounds and volatilised metals, to name a few. Fine PM dominates the surface area of particles in the atmosphere and it is thought that particle surface area is a better metric for assessing the ecological impact of particles due to their higher oxidising capacity [94].

Particulate matter of all sizes can be deposited on plant surfaces and exert both physical and chemical effects. Farmer [95] found that detrimental effects associated with particulate matter deposited on plants were most likely due to their chemistry rather than simply the deposited mass. Particles deposited onto plant surfaces can clog the stomata and reduce the rate of photosynthesis. If exposed for long enough, tissues and cells within the plant will die, and these dead tissues can fall out of the leaf leaving holes in the structure. For less severe exposure, plants become

discoloured, have weakened structures and stems, and leaves may become elongated or misshapen [18].

2.8 Legislation limiting pollutant levels in atmosphere

In recent decades, the issues of air quality and emissions from combustion of fuels in the residential space heating sector have become topics of considerable interest for legislating bodies, such as the European Union. Firstly, EU legislation relating to limiting source emissions was introduced. For example, the Large Combustion Plant Directive (LCPD) specifies that emissions from large combustion plants (>50 MW_{th}) must be continuously monitored and reported annually [11]. However, for small combustion installations (SCIs), i.e., those with a thermal input rating of less than 50 MW_{th}, no such legislation exists. Instead, quality standards relating to the performance of heating boilers specify limit values for carbon monoxide, organic compounds and total particulate matter according to different emission classes. For example, BS EN 15270:2007 [96] for pellet burners specifies five emission classes, the most stringent of which sets limits of 500 mg/m³, 50 mg/m³ and 30 mg/m³ for carbon monoxide, organic compounds and particulate matter respectively. The concentrations are referred to air at standard temperature and pressure, and a flue gas oxygen content of 10%. Standards for other types of heating appliance may specify different emission and/or performance values that the appliance must adhere to [97-100].

In addition to specifying emission limits for individual plants or appliances, limits on *national* emissions were also introduced, via

the National Emissions Ceiling Directive (NECD), implemented in 2001 [101]. This directive highlights the importance of reporting air pollutant emissions data for assessing progress in reducing air pollution in Europe and for ascertaining the compliance of member states with their commitments. The NECD deals with emissions of sulphur dioxide (SO₂), nitrogen oxides (NO_x), non-methane volatile organic compounds (NMVOCs) and ammonia (NH₃) emissions, and each party to the directive is limited to a specific emission level for each pollutant by 2010. One of the requirements that the NECD specifies is that member states must prepare and annually update their emissions inventory and projections, and submit them to the European Commission and European Environment Agency (EEA) by 31st December each year. Each of these reports should outline the final quantity of all pollutants for the previous year but one, and also contain provisional data for the previous year [101, 102]. The NO_x ceiling that was set for Ireland was 65 Gg. However, this was not achieved, with a mass of 90 Gg being reported in 2009 and 85 Gg having been predicted for 2010 [102]. The 2014 EEA Report lists the actual 2010 value as 76.7 Gg [132].

In addition, legislation pertaining to pollutant concentration in the atmosphere has been introduced which specifies emission limit values. These limits are generally specified as an average atmospheric concentration in units of µg/m³ with a number of permitted exceedances. In 1996, the Council of the European Union published the Air Quality Framework Directive (directive 96/62/EC), which aimed to improve air quality across Europe. This was implemented through a series of daughter directives, such as the

directives 1999/30/EC and 2000/69/EC [50, 103], which specified hourly and annual limit values for the concentration of oxides of nitrogen (NO_x), oxides of sulphur (SO_x), lead, and particulate matter (PM), as well as carbon monoxide (CO) and benzene in the atmosphere. These directives were superseded by the CAFE directive in 2008, [104], which specified that beginning on 1st January 2010, the average annual atmospheric concentration of NO_x in member states must be under 32µg/m³ and a maximum one-hour exposure of 140µg/m³ ought not to be exceeded more than 18 times a year. Similarly, for PM, the maximum one-hour exposure level is permitted to exceed 35µg/m³ 35 times a year and the specified annual average concentrations for PM₁₀ and PM_{2.5} are 28µg/m³ and 17µg/m³ respectively.

In Ireland, the Environmental Protection Agency (EPA) is responsible for licensing, enforcement, monitoring and assessment of activities associated with environmental protection. It is an independent public body that was established under the Environmental Protection Agency Act 1992 [105]. The functions of the EPA are as follows [106]:

- Environmental licensing
- Environmental law enforcement
- Resource and waste management
- Climate change regulation
- Environmental monitoring
- Environmental assessment
- Advice and guidance
- Water framework directive.

In addition, the EPA is also responsible for compiling emission inventory reports for submission to EMEP. Their annual informative inventory report (IIR), compiled

by the Office of Climate, Licensing and Resource Use (OCLR) within the agency, outlines the methodologies used to quantify the level of emissions and the sources of all data used [107]. The IIR is intended to inform Government departments, and other institutions, of the level of emissions, and is updated annually in accordance with the UNECE reporting guidelines [107].

2.9 Emissions measurements in the literature

A significant amount of research has been carried out on particulate matter emissions from wood pellet combustion. However, similar data for oil- and gas-fired boilers is sparser. Looking at wood-pellet boiler emission factors first, there is a clear mismatch between the value quoted in the EMEP guidebook and those found in the literature. EMEP suggest using an emission factor of 76 g/GJ for residential wood-pellet boilers, yet every other emission factor published in the literature suggests that for a modern pellet boiler the true PM concentration is a lot less than this. For example, Johansson *et al.* [108] measured particulate matter emissions from two modern wood-pellet boilers and found the concentration to be 16 g/GJ for the first and 13 g/GJ for the second boiler. They also examined intermittent low load operation for the same two boilers but the results are conflicting. In the case of the first boiler, reducing boiler output from a maximum of 22kW to 6kW caused a four-fold increase in PM emission factor, an expected result as combustion within the burner moves away from optimum conditions. However, when output was dropped further, this time to just 3kW, the measured PM emission factor was just 15 g/GJ, less than the concentration at

nominal output. When the same tests were performed on the second boiler, a different picture emerged. This time, when output was reduced to 6kW, the PM emission factor was 12 g/GJ, and when reduced further to 3kW, the concentration rose to 51 g/GJ. No explanation is offered in the article for this discrepancy. In the same study, it was found that the NO_x emission factor remained fairly constant for all tests averaging 67 g/GJ for boiler 1 and 62 g/GJ for boiler 2.

A study by Brunner *et al.* [109] examined pellet, wood-chip and log-wood boilers, and measured the concentration of PM produced for each of the fuels. For the pellet boiler, the PM concentration was found to be 13 mg/MJ, similar to the values published by Johansson *et al.* [108]. However, in this study, the authors did not examine the effect of varying load on the boiler, and the measured emission factor is based on steady operation. Wiinikka and Gebart [110] examined the effect of changing the primary air supply to an experimental burner had on the level of PM produced. They found that the PM emission factor varied between 4 g/GJ and 14 g/GJ, but once again, no investigation was made into the effect of changing the load on the boiler, or of modulating the fuel input. Of all the studies into pollutants from wood-pellet combustion, only one gave corresponding values for the NO_x emission factor. Johansson *et al.* [108] measured the concentration of NO_x for one of their boilers and found that under steady operation, the NO_x emission factor was 70 g/GJ, while during reduced operation, where the heat output has been reduced to 6kW and 3kW, the NO_x emission factor drops to just 64 g/GJ and 67 g/GJ respectively. As a comparison, the NO_x emission factor

suggested by the EMEP guidebook is 90 g/GJ.

Published data on NO_x and PM emission factors from oil and gas boilers is a lot sparser than for biomass fuels. Hays *et al.* [111] examined particulate matter emissions from combustion of heating oil in a residential oil burner. They found that most of the particulate matter released from the burner was in the form of soot from incomplete combustion and that its emission factor was 1.2 g/GJ. They did not measure NO_x concentration in their tests. Johansson *et al.* [108] measured PM and NO_x emissions from two residential oil boilers, as a comparison to the measurements taken from wood-fired heating units. They found that the two boilers had PM emission factors of 12 g/GJ and 6 g/GJ, and NO_x emission factors of 37 g/GJ and 41 g/GJ respectively. The NO_x emission factor suggesting in the EMEP guidebook for liquid fuels is 70 g/GJ. The PM emission factor they presented is very high, considering that the value published in the EMEP guidebook for use when calculating emission inventories is 3 g/GJ. The two boilers included in that particular study used older technology than is currently available, but even after taking that into account, the emission factors were very high.

There is no data available in the literature detailing PM emission factors for the combustion of natural gas. The quantities produced, even in rich combustion, are so low as to be almost negligible. The suggested values given in the EMEP guidebook are 70 g/GJ for NO_x and 0.5 g/GJ for PM.

2.10 Emissions from non-road transport

According to their most recent report detailing energy use in Ireland, the SEAI estimates that non-road transport (air, rail and marine) is responsible for approximately 5% of total primary energy use within Ireland [13]. When studying aviation activities, flights are divided into two activities. The first is landing and takeoff (LTO), which is defined as activities taking place below 3,000 feet altitude, while the second activity is cruise, defined as all activity above 3,000 feet. Aircraft engine manufacturers have to provide details of all engine operating parameters, including the total quantity of pollutants emitted. These emission factors can then be used to compile an emission inventory for aircraft. There are two flight types to consider when compiling emission inventories for the aviation sector: domestic and international. A domestic flight is defined as one which departs and arrives in the same country, while an international flight is one which arrives in a different country to the one it departed from. Emission inventories are written to a specific tier level, with Tier 1 simply looking at fuel use for domestic and international flights and calculating total emissions based on those figures. Using a higher tier leads to a more accurate inventory by examining flight patterns in more detail, for example, subdividing each flight into LTO and cruise modes.

Similar to aviation, the marine or navigation sector is split into domestic and international navigation. Marine engines are rated according to their speed (slow, medium and fast), with larger vessels using slower speed engines. All engines used for navigation must have their engines rated and

corresponding emission factors measured. The International Maritime Organisation (IMO) sets NO_x limits which depend on the size of the engine [112]. They have also set ambitious targets that, by 2016, all new vessels will produce 80% less NO_x than the current limit values permit [112]. The current Tier 1 method used for compiling emission inventories just looks at the total fuel consumed in the navigation sector, and uses an average emission factor to encompass all engine types. A higher tier approach, on the other hand, looks at engine size in more detail, and subdivides total fuel use into fuel used by each category of ship.

The final non-road transport sector is rail transport. In Ireland, two types of locomotives are in use: electric- and diesel-powered. Electric locomotives receive their power from the national grid and, hence, emissions associated with these vehicles are

accounted for under stationary combustion and the large combustion plant directive [11]. Diesel trains can be split into three categories: shunting locomotives, railcars and line-haul locomotives. Shunting locomotives are used for shunting wagons and have power outputs ranging from 200 kW to 2000 kW. Railcars are used for short distance rail traction and have power outputs of 150 kW to 1000 kW. Finally, line-haul locomotives are used for long distance transport and have a higher power output ranging from 400 kW to 4000 kW. Once again, a Tier 1 approach to compiling an emission inventory just looks at the total fuel used by the rail sector and combines it with emission factors measured by the engine manufacturers. A higher tier splits the total amount of fuel used into the proportional usage across the variety of different engine types.

3. Experimental Facility

As has been outlined, the aim of this research was to derive an improved emission factor for small combustion installations. In order to accurately determine such an emission factor, it was necessary to develop an experimental test facility where boiler performance at various operating loads could be tested, and corresponding emission concentrations measured. The alternative option of in-situ measurement of emissions from operational units was considered, but ultimately not pursued. It was deemed impractical due to technical issues with the sampling process, such as the difficulty in controlling the various boiler operating parameters.

In order to estimate real-world emissions from domestic installations, it was necessary to use a representative duty cycle in the test facility. Data loggers were used to monitor flue gas temperature in a number of installations in both urban and rural locations. From the resulting flue temperature-time trace, it was possible to determine actual running time, the total number of start-ups, transients, etc. Once sufficient data had been gathered, an accurate duty cycle could be derived and applied to the test facility, so that accurate measurements of associated real-world emissions could be obtained.

In a typical installation, the boiler keeps running until the water flow temperature exceeds the set point on the thermostat, at which point it switches off. When the water temperature falls below a lower threshold, the boiler switches back on and thereafter runs intermittently, maintaining the flow

temperature at high temperature, typically 70-80°C. The actual boiler running time depends on a number of factors, such as ambient temperature, heat loss/gain from the building to the atmosphere, user comfort levels, etc.

Three types of domestic scale boiler were tested, wood-pellet, oil-fired and gas-fired. [Figure 3.1](#) shows the outline plan for the test facility. Initially, the wood-pellet boiler was installed and the test facility commissioned. In later tests, the pellet boiler could be exchanged for either oil- or gas-fired boilers. A Gerkros wood-pellet boiler with a nominal thermal input of 20 kW was chosen as an example of a boiler that was available on the Irish market. The pellets are top-fed from a hopper through an auger into the combustion chamber. The boiler heat output is modulated to one of five discrete power levels, by varying the pellet feed rate; the speed of the fan supplying air to the combustor adjusts automatically to match the pellet feed rate. The pellets are fed into the burner and heated with an electric element until ignition, after which the combustion fan turns on until the flame is fully developed and the boiler then operates at full power.

In order to simulate a domestic load on the boiler, the unit was plumbed into a water-to-air heat exchanger with a variable speed fan. By increasing the fan speed, a higher cooling rate was applied to the hot water flowing in the circuit, simulating operation in a dwelling on a cold day. Conversely, running with a lower fan speed reduces this cooling effect, mimicking operation on a warm day. A

mixing valve was also included in the plumbing circuit which could be used to divert some of the hot water from the flow side back to the return side. This had the net

effect of reducing the cooling rate further and, hence, was used when operation at very low load was under consideration.

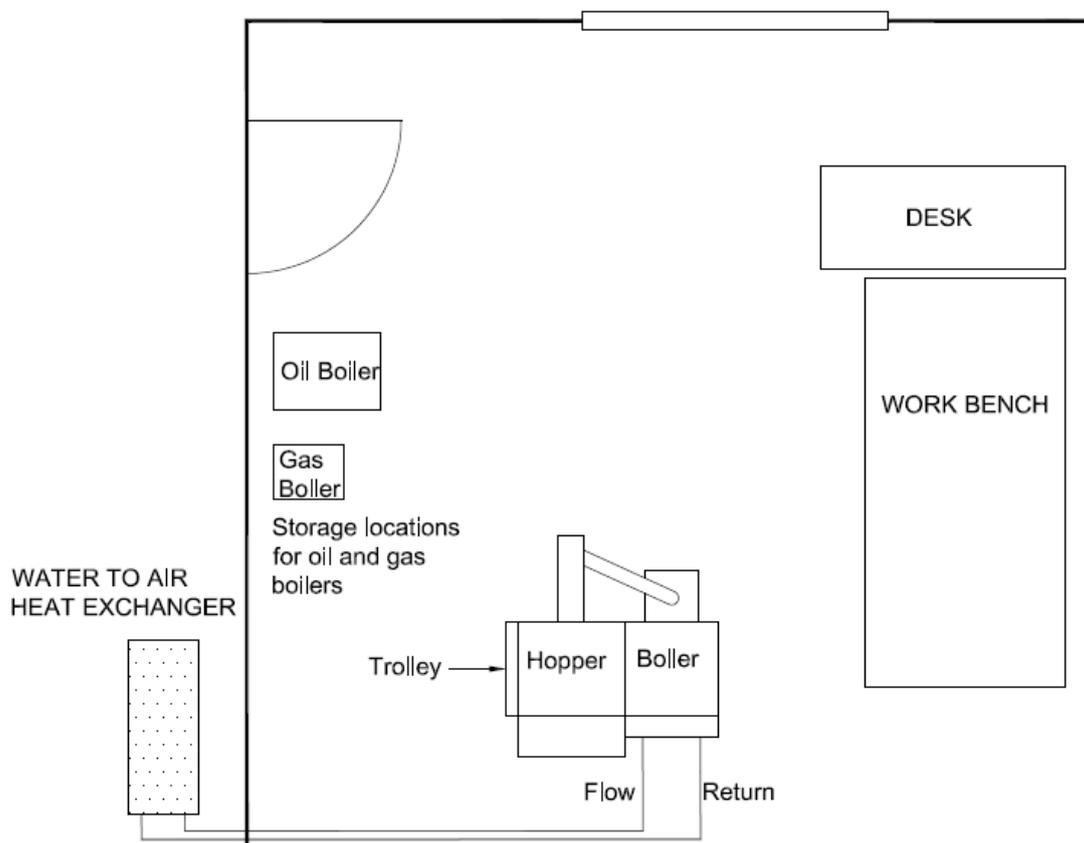


Figure 3.1. Outline plan of test facility

A common plumbing circuit and measurement system was used for all three boilers. A schematic of the measurement section on the plumbing circuit is shown in [Figure 3.2](#). The parameters that were

measured were water flow and return temperatures, flue temperature, ambient temperature, water flow rate, cooling fan speed, mixing valve position, heat output and fuel consumption rate.

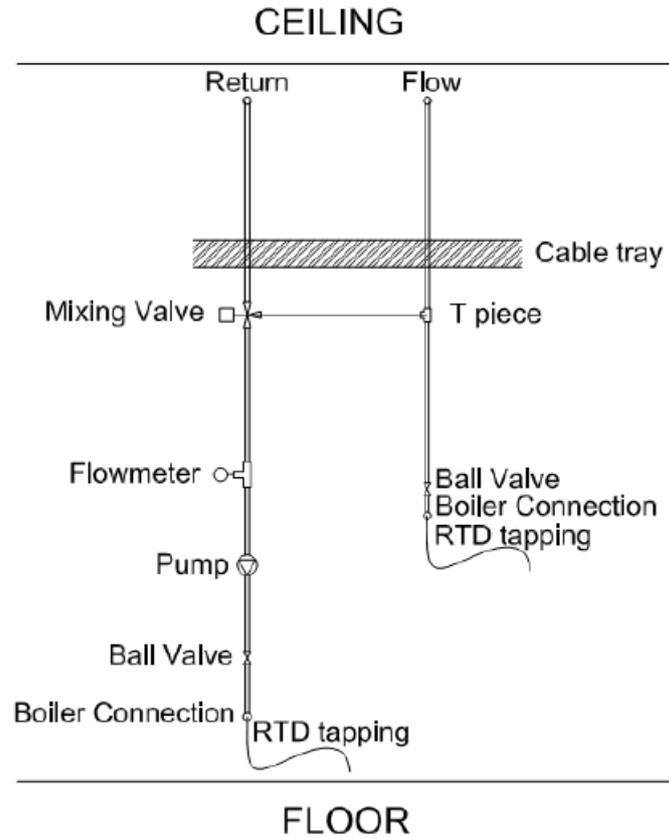


Figure 3.2. Measurement section of plumbing circuit

Boiler performance was recorded and monitored using a National Instruments NI cDAQ-9172 data acquisition system (DAQ), shown in [Figure 3.3](#). This USB chassis had a number of plug-in modules for the various measuring devices used. The measurement signals were recorded, and displayed on

screen using National Instruments LabVIEW software. LabVIEW is a graphical programming language used for developing complete instrumentation, acquisition and control applications, and allows the user create a completely customisable DAQ.

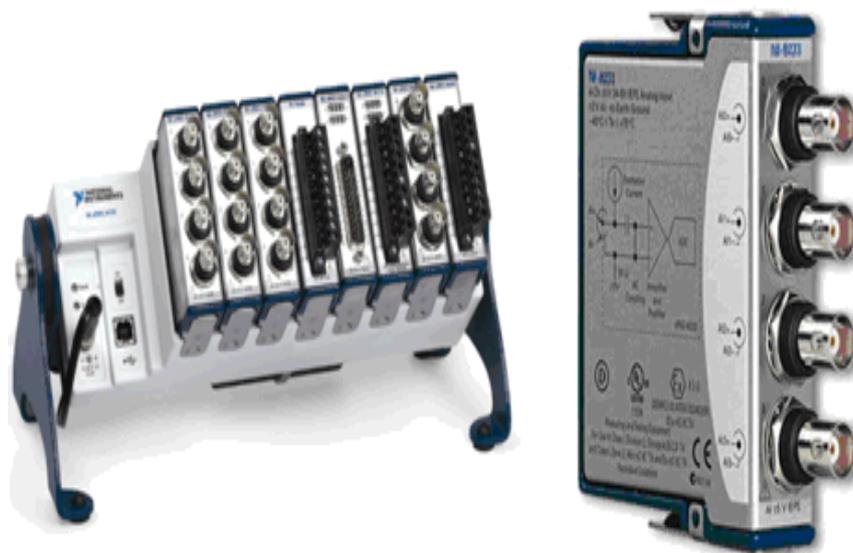


Figure 3.3. NI cDAQ-9172 chassis. Source: [113]

Using the programme, the user creates a virtual instrument (VI) with a user interface known as the front panel. On this front panel, the user places virtual indicators, gauges, switches, buttons, etc., to display the data that is being measured. The front panel is controlled by an executable programme called the block diagram. The block diagram

has a similar appearance to a wiring diagram or circuit board. Virtual wires are used to connect the various components on the front panel to indicate the flow of data between the various objects. A sample front panel and block diagram are shown in [Figure 3.4](#) and [Figure 3.5](#) respectively.

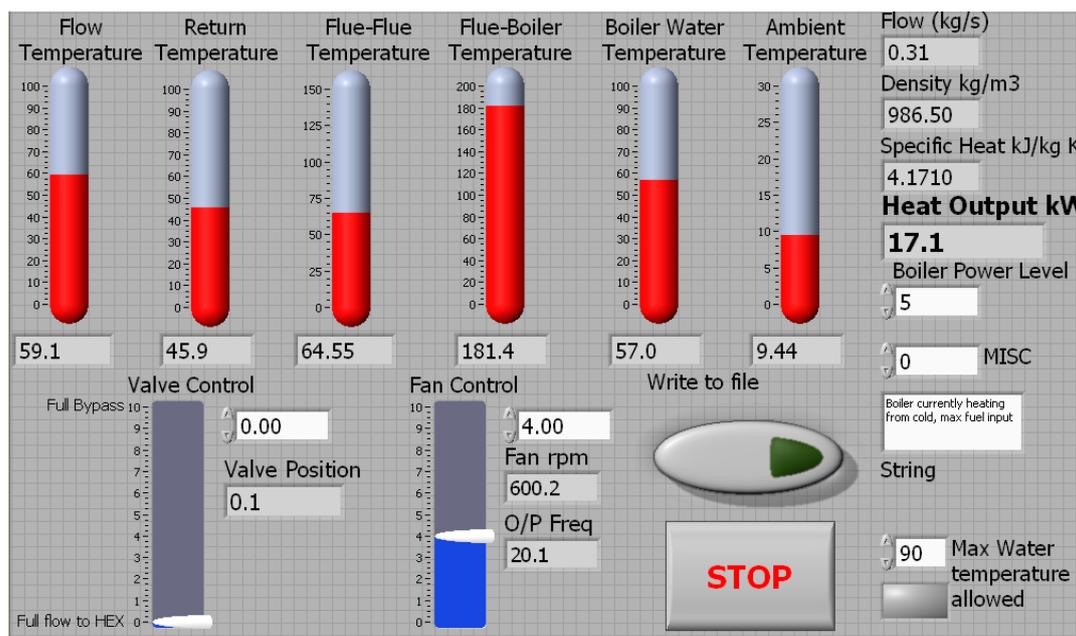


Figure 3.4. Sample LabVIEW front panel

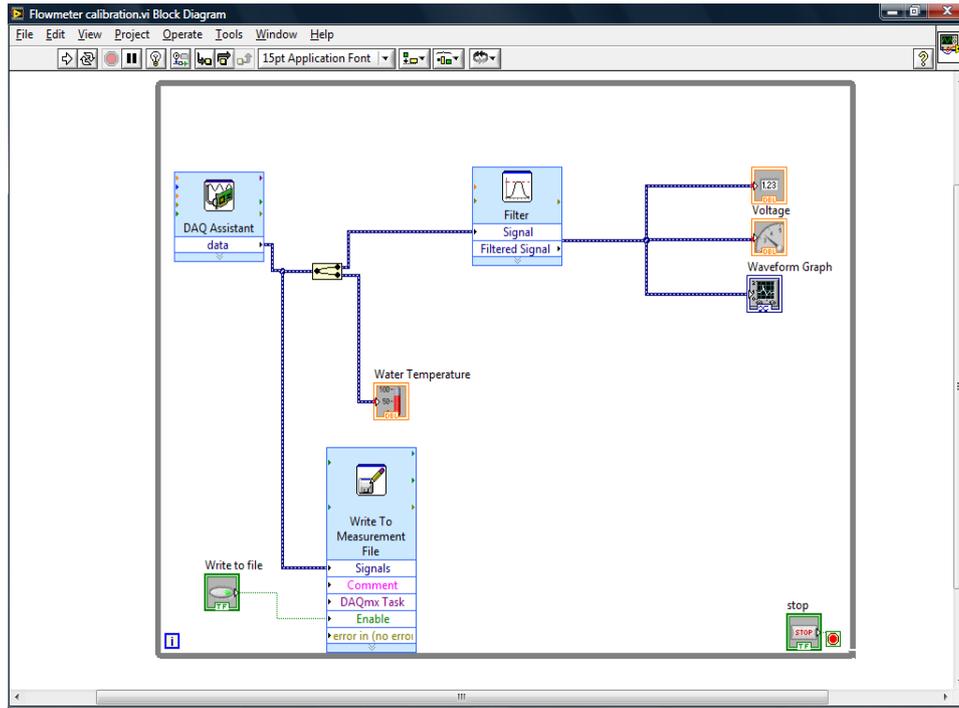


Figure 3.5. Sample LabVIEW block diagram

Water flow and return temperatures were measured 4-wire RTDs, ambient temperature using a J-type thermocouple and all other temperatures with T-type thermocouples. RTDs were chosen for their higher accuracy limits, as water temperatures were the critical parameters for assessing boiler performance. The water flow rate was measured using an Icenta FM10 turbine flowmeter. The principle of operation is that, as water flows through the meter, it causes a freely supported rotor to spin, and the speed at which it does is directly proportional to the flow rate. This generates a frequency at 5V, which is fed into a frequency-to-voltage converter and then into the DAQ analogue input chassis module.

A Testo 350 M/XL analyser was used to continuously measure the concentrations of O₂, CO, CO₂, NO_x and H₂, while particulate matter concentration and size distribution were measured using a Dekati Low Pressure

Impactor (DLPI). The Testo unit uses a series of electrochemical cells to measure the concentrations of O₂, CO, NO_x and H₂ and a non-dispersive infra-red (NDIR) sensor for CO₂. The measurement principle used in the electrochemical cells is that of ion selective potentiometry. When the flue gas enters the cell, it chemically reacts with an electrolytic matrix and releases ions which change the voltage across the electrode, and this voltage is proportional to the concentration of gas present [114].

In comparison to measurement of gaseous emissions, such as NO_x or CO, particulate matter measurement is considerably more challenging. As PM is made up of a 'complex mixture of organic and inorganic substances' [44], and contain particles in different physical states, a change in the measuring principle used to measure PM concentration will lead to different results. Depending on the jurisdiction under which boilers are tested, different standards will apply. The

European and International standard for measuring particulate matter concentration is BS EN ISO 9096:2003 [115]. This standard defines particulate matter as 'all compounds that may be collected by filtration under specified conditions' but does not include condensable compounds or reaction products that form at temperatures lower than the sampling temperature [115]. The standard specifies that the sampling temperature be either 160°C or the temperature of the gas at the sampling point.

In contrast, the United States Environmental Protection Agency (USEPA) Method 5 specifies a two-stage measurement process, which quantifies both the solid particles as well as the condensable fractions contained within the aerosol [116]. Consider [Figure 3.6](#) which highlights, qualitatively rather than quantitatively, the effect different sampling methods have on the measured concentration. The 'SP' method follows the steps outlined in the BS EN ISO 9096:2003 standard while the 'SPC' method follows the USEPA Method 5 process, yielding a different result. Hence, it is important, when performing emissions measurements on a boiler, to specify to which standard the tests are being performed in agreement with, because if two different methods have been used, then the results are not directly comparable.

In this project, PM mass was measured using the method given in the BS EN ISO 9096:2003 standard [115]. The measuring instrument used was a Dekati Low Pressure Impactor (DLPI). This cascade impactor operates on the principle of inertial impaction. It separates the aerosol into 13 different size fractions and captures them on specially prepared, pre-weighed aluminium substrates. The particles are sampled isokinetically from the flue and passed through an insulated sample line to the DLPI. Isokinetic sampling is a procedure to ensure that a representative sample of aerosol enters the inlet of a sampling tube when sampling from a moving aerosol stream [36]. It is very important that this criterion is met, particularly for larger particles, because otherwise it leads to errors in the calculated mass concentration and a distortion in the size distribution. When particles enter the impactor, they are accelerated through a 90° bend (see [Figure 3.7](#)). Large particles, with an aerodynamic diameter greater than the stage cut point, are unable to make the turn due to their own inertia and, instead, are impacted onto a substrate. These substrates are analysed gravimetrically before and after sampling to give the particulate mass concentration and size distribution.

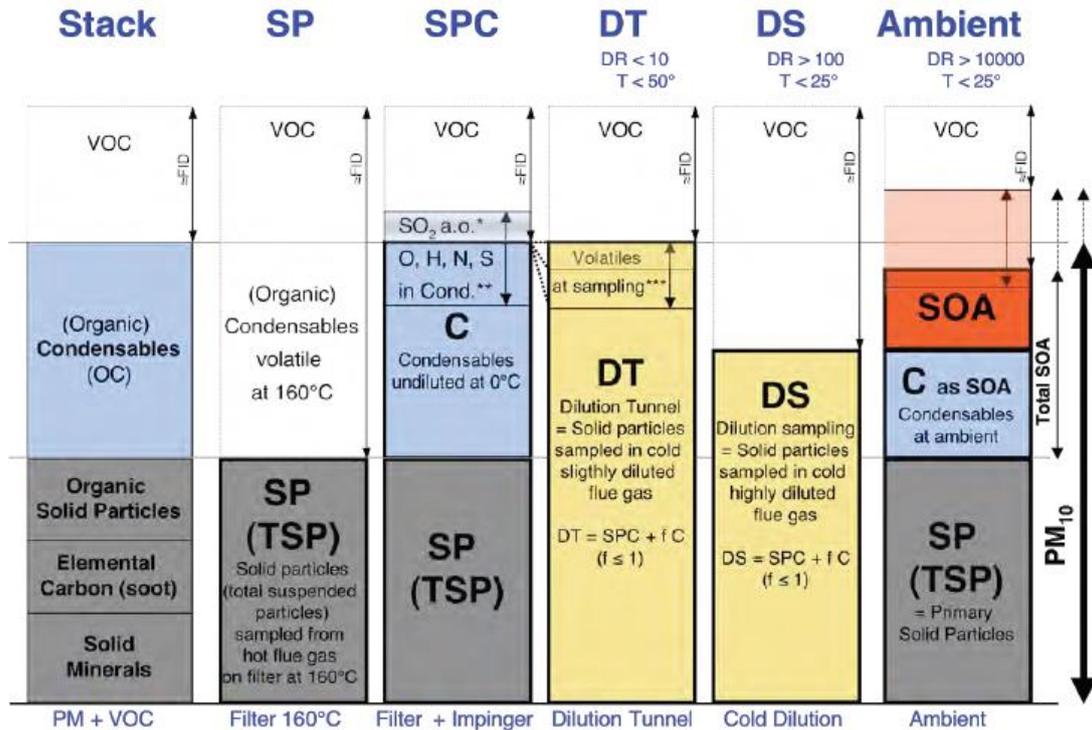


Figure 3.6. Comparison of PM sampling methods with ambient PM. SP: Filter sampling to collect solid particles; SPC: Filter and impinger sampling which collects solid particles and condensables; DT: Dilution tunnel with typical dilution ratio in the order of 10 resulting in a PM measurement including SPC and most or all of C. Hence, DT is identical or slightly smaller than SPC+C due to potentially incomplete condensation depending on dilution ratio or sampling temperature; DS: Dilution sampling with high dilution ratio (>100); PM₁₀: Total particulate matter <10µm in the atmosphere including SP and SOA; SOA: Secondary organic aerosols, consisting of condensables C at ambient and SOA formed by secondary reactions such as photochemical oxidation. Source: [117, 118]

However, particle bounce is a significant problem when using impactors. It occurs when particles fail to adhere to the collection substrate and instead bounce off and are collected on the impactor's side walls or may not be captured again at all. Hence, it can lead to large uncertainties in both the overall mass concentration and the size distribution. To improve the capture efficiency of the stages, a thin layer of vacuum grease (Apiezon L) was applied to the surface of the

aluminium. The grease was dissolved in Toluene at a dilution ratio of 20:1 and applied with a brush to the substrates. These were then baked at the sampling temperature for two hours to volatilise the residual toluene, so as to avoid mass losses during sampling. The grease was applied as a very thin layer, typically <2µm, as any more causes grease to be blown off the stage by the high jet velocities.

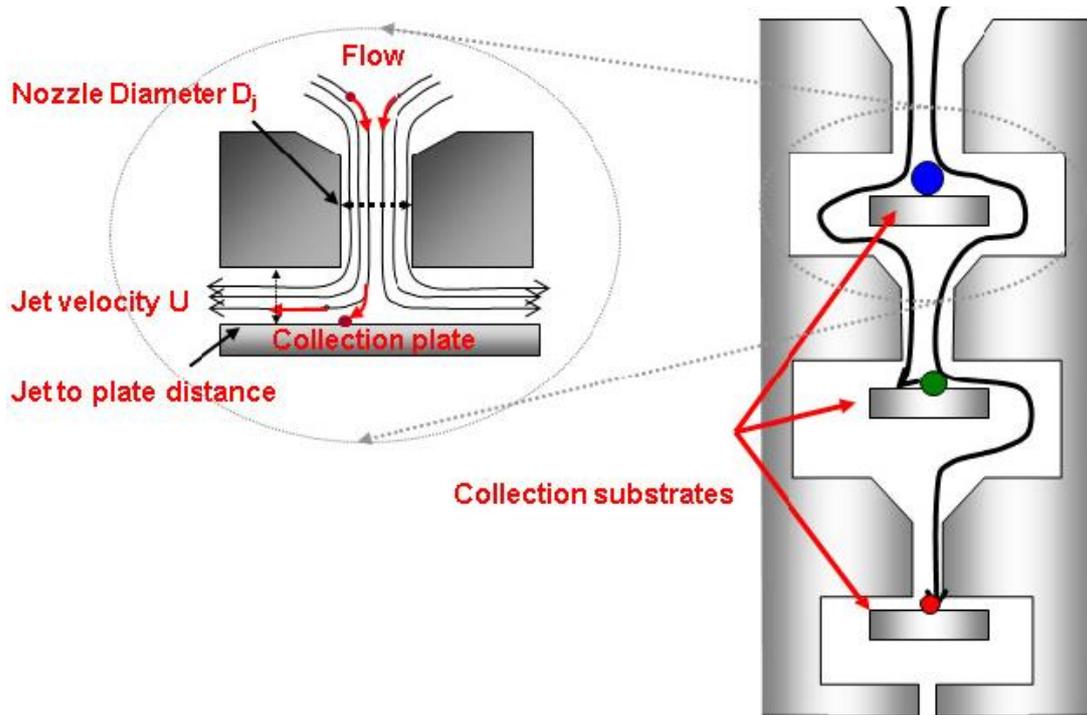


Figure 3.7. Dekati Low Pressure Impactor (DLPI) operating principle [119]

Another hurdle that must be overcome when sampling particulate matter is the issue of water vapour condensation. As the flue gas cools as it is being transported from the flue to the impactor, the water vapour will begin to condense. This can wreak havoc with the measurement process because when the water condenses inside the impactor, onto the impaction substrates, it can wash some of the collected particles off, and the added water nullifies the test as the substrate mass will have changed due to both particles and water gathering on the surface. There are two options to overcome this issue: one is to use a heating jacket, the other is to use a sample dilution system. Dilution, on the other hand, offers a number of benefits. First of all, when sampling high concentration aerosols, such as that from wood-pellet combustion, it reduces the concentration and allows the test to continue for a longer period of time. Additionally, when the gas cools, the water vapour can condense out and be captured in

a water trap rather than on the PM collection substrates within the impactor. By using a high dilution ratio, it is possible to cool the flue gas very rapidly. This has the effect of halting any further particle growth and agglomeration, and allows for a better examination of the size and concentration of particles as they leave the boiler. When sampling without dilution, by the time the particles reach the impactor, they will have grown significantly due to agglomeration and condensation of vapours onto their surfaces, and the measured size distribution can be quite different to the true distribution. Due to the high quenching rate, this method also captures the vapours in the flue gas, which would eventually form secondary aerosols after being released to the atmosphere. However, for this project, a heating jacket was chosen, as it was a less expensive and less complicated solution than the sample dilution. The impactor was enclosed in the heating jacket whose temperature was set to

the same value as the flue gas at the sampling point, and above the dew point temperature for the flue gas. Using this method, it was only possible to measure the mass of solid particles released from combustion and none of the condensable fraction at all. Use of a diluter could have given more accurate detailed information but was not used this time due to budgetary constraints.

Use of the DLPI poses considerable challenges for the user. It is desirable to sample for as long as possible, so as to gather a mass of PM that the microbalance can actually detect. However, the unit is very sensitive to particle bounce and it takes a lot of trial and error to find the optimum sample time for a particular test condition. The application of the grease layer to the substrate is also very important. It is critical that the layer is very thin and has a uniform thickness across its entire area. If not, then the high velocity air jets will simply blow grease off the surface, particularly when that grease has been heated to avoid water condensation. Once grease blow-off, or particle bounce occurs, the test becomes null and void and must be repeated.

Because of these issues with the DLPI, a filter sampling system was developed to be used in a parallel test series on the oil- and gas-fired boilers. These tests were not necessary with the wood-pellet boiler because, due to the much higher PM concentration, it was much easier to find the optimum sample time before particle bounce became a problem. The filter system was limited in the sense that it could only give mass concentration information, and not size distribution, but it did make the sampling process much simpler as the problems of

particle bounce and grease blow-off did not arise. Hydrophobic PTFE filters with a diameter of 47mm and a pore size of 0.2 μ m were used in conjunction with an inline filter cassette holder. A hydrophobic filter is one which will not wet in water [120]. This was an important property for sampling because, as the flue gas cooled in the sample line, water vapour condenses out and could damage a filter that does wet in water, i.e., a hydrophilic filter. An advantage of the filter sampling system was that it only required one collection filter to be weighed before and after the sample. In contrast, when sampling PM using the DLPI, 13 separate collection substrates had to be weighed, and often the mass collected on a stage was so small it was almost immeasurable. Hence, the contribution of the microbalance uncertainty to the overall emission factor uncertainty was higher when sampling with the DLPI.

For this reason, as well as the problems and uncertainties associated with the DLPI, the emission factor calculated using the filter sampling system is taken as the definitive emission factor for the oil and gas boilers under a particular test condition, and the DLPI data is used solely as a source of size distribution data. Initially, tests were carried out at each condition using the DLPI and then the test conditions were replicated and particulate matter sampled using the filter sampling system. It is important to note that DLPI and filter samples came from different tests, and that while every effort was made to ensure test conditions remained identical, certain variations did occur.

A picture of the boiler test facility, with the wood-pellet boiler installed, is shown in [Figure 3.8](#).

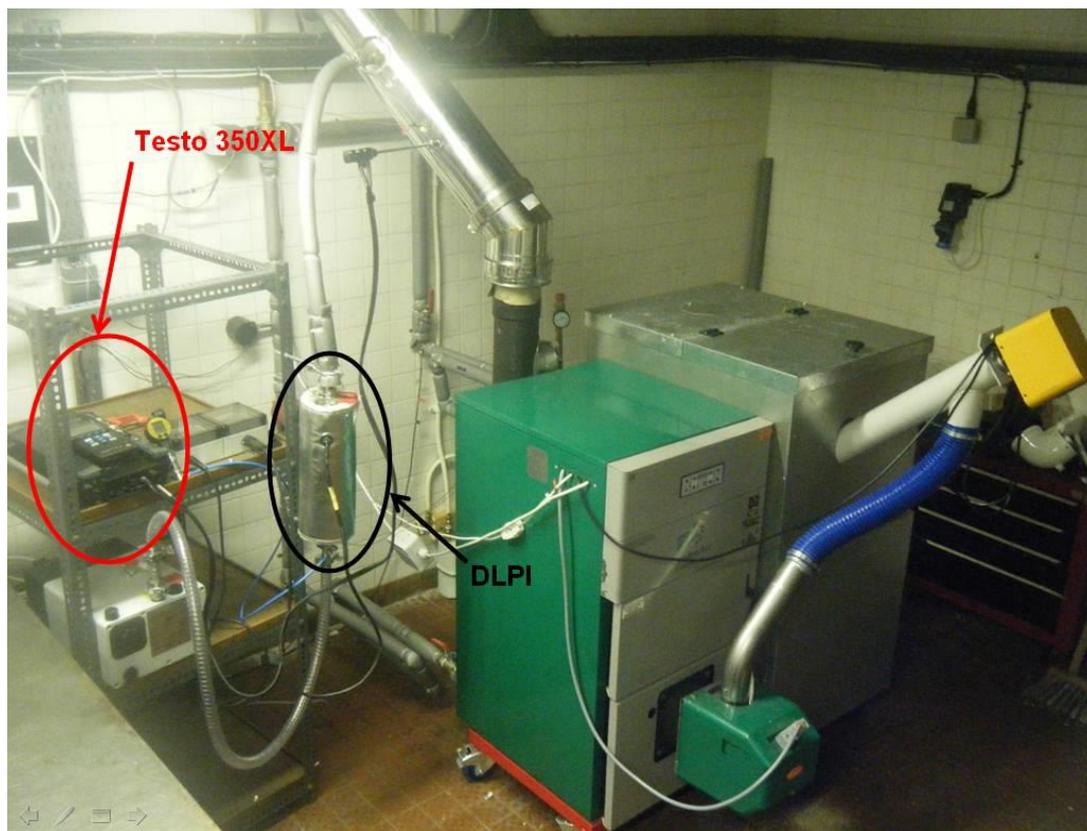


Figure 3.8. Wood-pellet boiler installed in test facility

Measurement of the rate of fuel consumption is a critical requirement for the testing series. For the gas boiler, a Krom Schroder BK-G4 flow meter was installed. The BK-G4 meter is the same model as Bord Gais use when connecting a gas supply to a house. The meter consists of four measuring chambers separated by a diaphragm, which are filled and emptied periodically. The movement of the diaphragm drives a counter via a gear set. The volume of gas used during a test is found simply by taking the volume reading off the meter at the start and end of the test, with the difference being the volume of gas consumed (m³). The oil consumption was measured using a fuel gauge with a volume of 800ml. The time taken to empty the fuel gauge was recorded, and once empty, the fuel gauge refilled and the boiler kept

running. The time taken each time was recorded and an average fuel flowrate value for the whole test calculated. Measurement of the consumption of wood-pellets was less straightforward. As it was a solid fuel, there was no continuous flow to measure. Instead, the feed auger operated for varying lengths of time, depending on the power level the boiler was operating at. The feed auger was calibrated to measure the mass of pellets delivered each fuel feed and, using this, the total mass of pellets consumed could be found.

There are two methods used to measure the efficiency of a heating boiler. The first is to measure the heat output, using the following equation:

$$\dot{Q}_{out} = \dot{m} C_p \Delta T$$

Equation 7: Formula to calculate boiler heat output

where \dot{m} denotes the flow rate of water in the heating circuit, C_p the specific heat capacity of the water, and ΔT the temperature difference between the flow and return. The rate of energy input to the boiler can then be

found by multiplying the mass flow rate of fuel by the calorific value. Once these two terms have been found, the efficiency is found using the following equation:

$$\eta = \frac{\dot{Q}_{out}}{\dot{Q}_{in}}$$

Equation 8: Boiler Efficiency

This method is used to calculate the efficiency values for all tests carried out in the laboratory. Alternatively, the efficiency can be calculated using the 'loss method'. Using this, the rate of heat loss by various mechanisms is calculated and taken from 100%. Heat loss through the flue is the main source, and its value depends on the temperature, flow rate and composition of the flue gas. This method is used during the field testing campaign, where it was not possible to measure the water flow rate or temperature while it was circulating. The efficiency was based on the lower calorific value for the wood-pellet and oil boilers, and the higher calorific value for the gas boiler. The calorific value of a fuel is the amount of heat released when a unit amount of fuel at

room temperature is completely burned, and the combustion products are cooled to room temperature [20]. The higher calorific value is used if the water in the flue gas is in liquid form, whereas when the water vapour in flue gas is in vapour form, the lower calorific value is used. The lower calorific value was used on the pellet boiler, as it was not designed to condense water. The oil boiler did have a condensing element, but rarely ran in condensing mode during tests, so the lower calorific value was used on it as well. The gas boiler on the other hand spent long durations operating in condensing mode and so the higher calorific value is used for that particular boiler. For boilers tested in the field measuring campaign, all efficiencies are based on the lower calorific value.

4. Experimental Programme

As outlined in the Executive Summary, there were three strands to the testing and measurement programme. The initial step was to monitor a number of installations, to determine a representative duty cycle that could be used in the test facility. The second strand was to use these duty cycles in the laboratory, where boiler performance and emissions concentration could be accurately measured. The final strand was to conduct a field measuring campaign. For reasons outlined in [Section 1.3](#), it was not possible to accurately measure boiler performance in a domestic installation, as there were too many variables that could not be controlled, and in addition no PM data could have been measured, as the DLPI used was not portable. So the field measuring campaign was designed to complement the laboratory testing by examining how emissions from a real-world boiler, (NO_x in particular) of varying ages and service levels, differed from those measured in the tightly controlled, modern, state-of-the-art boilers used in the laboratory.

In order to determine a representative duty cycle to be used in the test facility, twelve Hobo data loggers and J-type thermocouples were used to monitor flue gas temperature at one minute intervals at a variety of locations. The resulting time-temperature traces were then analysed to find the total running time, typical on-off duration, number of start-ups and transients, etc. A total of six oil boilers, four gas boilers and three wood-pellet boilers were monitored over a period of time, ranging from two weeks to ten months.

When selecting installations to monitor, the aim was to have a good spread across different location types, dwelling types, boiler age, service history, etc. Of the six oil boilers that were monitored, three were in urban locations and three in rural. There was a good spread from brand new boilers to old models with a poor service history. Information on the six monitored installations is given in [Table 4.1](#). The sampling period ranged from two to six months.

Table 4.1. Details of the six monitored oil-fired installations

	Location	Type	Boiler Size [kW]	Notes on Installation
1	Rural	Det. Bungalow	44	Modern condensing boiler, new house
2	Rural	Det. Dormer	35	3-year-old, non-condensing boiler. Well insulated house.
3	Rural	Det. Bungalow	29	Old boiler, >20 years old, old house, poorly insulated
4	Urban	Det. Dormer	32	Non-condensing boiler, <10 years old
5	Urban	Det. Dormer	32	Non-condensing boiler, <10 years old
6	Urban	2 storey, semi-det.	30	Old (~25 years) boiler

Similarly, when choosing gas installations to monitor, the intention was to match the actual distribution of gas boilers across Ireland. As gas is normally only available in urban areas, all four monitored installations

were in an urban setting. Once again, they ranged from new high efficiency condensing boilers to older units in less efficient houses. The details of the four monitored gas-fired installations are given in [Table 4.2](#).

Table 4.2: Details of the four monitored gas-fired boilers

	Location	Type	Boiler Size [kW]	Notes on Installation
1	Urban	Terrace	15	Old house, low insulation levels, boiler >10 years old
2	Urban	Semi-det.	20	Well insulated house, boiler >10 years old
3	Urban	Semi-det.	29	Modern house, high insulation levels, new condensing boiler
4	Urban	Semi-det.	26	Old house, upgraded insulation levels, new non-condensing boiler

Finally, details of the wood-pellet boilers that were monitored are given in [Table 4.3](#). Unfortunately, as these boilers are a new technology in Ireland, and to date total sales

have been low, it was difficult to find a large number of installations that were suitable for monitoring. Therefore, all three boilers that were monitored were Gerkros models, the

same as that used in the laboratory. The first two were an upgraded version of the laboratory model. The combustion technology and control system were the same, but an enhanced user control

interface had been developed and the maximum heat output increased from 20kW to 25kW. The third boiler was the same model as the laboratory installation.

Table 4.3. Details of the three monitored wood-pellet boilers

	Location	Type	Boiler Size [kW]	Notes on Installation
1	Rural	Detached	25	New house, well insulated, boiler <3 years old
2	Rural	Detached	25	New house, well insulated, boiler <3 years old
3	Rural	Detached	20	New house, well insulated, boiler <3 years old

The second strand in the process was to use the results from the duty cycle analysis on a boiler installed in a test facility, where emissions concentration at all operating conditions could be measured accurately. A comprehensive series of tests, designed to test every aspect of boiler operation was performed on the wood-pellet, oil and gas boilers, and the PM and NO_x emission factors were measured during each of these tests. For the wood-pellet boiler, steady state tests were performed at each of the five power levels, with particular emphasis on power level 5, as this is the most common operating condition. In addition, the entire start-up phase was also replicated in the laboratory, and PM and NO_x emission factors measured.

For the oil boiler, three operating conditions were under examination: steady state, cold start and warm start. Cold start was defined as the first ten minutes of operation after being switched off for an extended period of

time. In this state, the boiler and heating system are at ambient temperatures and require significant energy input to raise the system temperature up to the levels found during steady state condition. Steady state condition was defined as the running time after the boiler had reached steady temperature to the point where it shut down due to the temperature exceeding the thermostat set point. Warm start was defined as the intermittent operation typically found in oil and gas boilers, whereby the system cycles on and off in order to maintain the temperature as close to the thermostat value as possible. Two methodologies were used to measure the PM concentration. In addition to the DLPI, a separate filter sampling system which could measure the overall mass concentration was developed and used, as it overcame a lot of the problems posed by measurement with the DLPI. For these tests, the DLPI was used to measure the mass size distribution only, with the

actual emission factor being calculated from the filter sample tests.

The gas boiler tests followed a similar pattern to the oil samples, with steady state, cold and warm starts being under investigation. However, in this case, the extremely low levels of PM meant that the concentration could not be measured by the DLPI and instead the filter system was used for all tests.

As an additional tool, a scanning electron microscope (SEM) was used on selected particle samples from the wood-pellet and oil boilers to examine particle morphology and composition. Particles from the gas boiler were not examined, as the concentration was so low that not enough particles could be captured for examination. The SEM can be used to image very tiny particles down to 1 nm in size. In addition to imaging the particles, the microscope can also be used to find the composition of those particles, using a process called energy dispersive x-ray spectroscopy (EDX). EDX gives reliable

information about the elements present but is not very accurate in determining the relative amount of each element. Some images of particles found from combustion of wood pellets and kerosene are included in the Results section.

A field measurement campaign was carried out to complement the laboratory testing programme. A total of 31 domestic boilers (23 oil-fired, six gas-fired and two wood-pellet) and 10 commercial/light industrial boilers (four oil-fired, five gas-fired, one wood-chip) were sampled. It was not possible to measure the PM concentration from these boilers, as the DLPI and filter sampling system were not portable. In addition, PM sampling required both close control on all operating parameters so as to ensure constant repeatable test conditions as well as knowledge of both the fuel and flue gas flow rates, none of which were available in the field testing campaign. Details of the 31 domestic installations that had their flue gas sampled are given in [Table 4.4](#).

Table 4.4. Details of domestic scale boilers sampled as part of the domestic field measurement campaign

Sample No.	Location		Fuel	Boiler Size [kW]
1	Kildare	Rural	Oil	20
2	Kildare	Rural	Oil	35
3	Kildare	Rural	Oil	35
4	Wicklow	Urban	Oil	32
5	Wicklow	Urban	Oil	32
6	Kildare	Rural	Oil	35
7	Kildare	Rural	Oil	35
8	Kildare	Rural	Oil	25
9	Kildare	Urban	Oil	19
10	Kildare	Rural	Oil	44
11	Kildare	Urban	Oil	19
12	Dublin	Urban	Oil	26
13	Mayo	Urban	Oil	20
14	Mayo	Rural	Oil	20
15	Dublin	Urban	Oil	30
16	Kildare	Rural	Oil	23
17	Dublin	Urban	Oil	20
18	Kildare	Rural	Oil	35
19	Kildare	Rural	Oil	35
20	Kildare	Rural	Oil	35
21	Kildare	Rural	Oil	35
22	Kildare	Rural	Oil	35
23	Kildare	Rural	Oil	25
24	Dublin	Urban	Gas	15
25	Dublin	Urban	Gas	20

26	Dublin	Urban	Gas	25
27	Dublin	Urban	Gas	26
28	Dublin	Urban	Gas	11
29	Dublin	Urban	Gas	15
30	Dublin	Rural	Wood Pellet	25
31	Wicklow	Rural	Wood Pellet	25

Details of the boilers sampled as part of the commercial scale field measurement campaign are given in [Table 4.5](#).

Table 4.5. Details of the ten commercial scale boilers sampled as part of the field measurement campaign

Sample No.	Location	Building Type	Fuel Type	Boiler Size [kW]
1	Monaghan	Office Building	Oil	120
2	Monaghan	Office Building	Oil	120
3	Mayo	Office Building	Oil	332
4	Mayo	School	Oil	250
5	Dublin	Office Building	Gas	65
6	Dublin	Office Building	Gas	145
7	Wexford	Office Building	Gas	500
8	Kilkenny	Office Building	Gas	250
9	Kilkenny	Office Building	Gas	310
10	Wexford	Office Building	Wood Chip	400

Improved Emissions Inventories for NO_x and Particulate Matter from Transport and Small Scale Combustion Installations in Ireland (ETASCI)

Sampling involved turning on the boiler, allowing it to reach a steady output and then switching on the sampling equipment for 10-15 minutes and recording the flue gas

temperature and composition. [Figure 4.1](#) shows examples of boiler that were sampled as part of the field measuring campaign.



Figure 4.1. Field sampling of commercial (left) and domestic (right) boilers

5. Results: SCI

5.1 Introduction

This chapter presents the results from the testing programme carried out on the wood-pellet, oil and gas boilers, as well as results from the field testing campaign. The chapter is divided into a number of different sections. [Section 5.2](#) details the investigation and measurement of typical duty cycles for boilers installed in the field. Next are [Sections 5.3](#) and [5.4](#), which describe PM and NO_x emission factor measurements carried out in the laboratory. [Section 5.5](#) shows the results of microscopy performed on a selection of collected particles. Finally, [Section 5.6](#) describes the results obtained from the field testing campaign that was carried out. It was not possible to measure the particulate matter concentration during this campaign, as the low pressure impactor used was not readily portable, and accurately measuring particulate matter emission factors required very tightly controlled test conditions, something which was not possible outside the laboratory.

5.2 Duty cycle test results

As outlined earlier, the approach taken in the present study to determine suitable emission factors for domestic heating boilers involved two steps. The first step was to monitor a range of installations to determine a suitable duty cycle. In the second step, these duty cycles could be replicated in an experimental facility and the relevant emission factor measured accurately under controlled conditions. The data gathered during the first step is presented in this section. The data

obtained from the duty cycle analysis can then be coupled with laboratory measured emission factors and a weighted average emission factor to be used when compiling emission inventory reports can be calculated.

The analysis of the oil boiler duty cycles is presented in [Section 5.2.1](#). [Section 5.2.2](#) describes the results of duty cycle measurements on gas boilers, while the results from the monitored wood-pellet boilers are given in [Section 5.2.3](#).

5.2.1 Oil boiler duty cycle analysis

Typically in Ireland, boilers tend to be oversized and houses have lower insulation levels, when compared to corresponding buildings in Eastern Europe or Scandinavia. As a result, boilers tend to get up to temperature relatively quickly and then enter an intermittent firing sequence, whose aim is to maintain system temperature within a particular operating window. A firing cycle for a boiler is now described.

When the boiler is switched on, the temperature of the whole heating system is at ambient levels. The boiler then runs for an extended period of time until the system reaches a set temperature, determined by a thermostat. At this point, the boiler switches off. Gradually the system temperature falls as heat is transferred to the house, and once it drops below a certain threshold value, the boiler switches back on again and runs until the temperature reaches the thermostat

setting again. This process repeats as long as the boiler is switched 'on'.

Typically in Ireland, operation of oil boilers is characterised by three distinct phases:

1. Cold start: This is taken as the first ten minutes of the initial heating period. After this initial warming phase, the combustion chamber in the boiler is at working temperature and any further temperature increase is at a diminishing rate.
2. Steady state operation: In this phase, the boiler operates at maximum output until the system temperature reaches the thermostat set point.
3. Warm start: This final phase is characterised by relatively high frequency intermittent operation, designed to maintain boiler output at a constant level.

This operating mode is characteristic of oversized boilers. Where boiler output more closely matches the demand for heat, the boiler will run at steady state output for longer and the intermittent warm start does

not feature as strongly in the duty cycle. By analysing these three phases, it was possible to determine relevant information, such as cold start duration, typical on-off times, etc.

As an example, consider [Figure 5.1](#), a typical time-temperature trace for a domestic oil-fired boiler. The plot was taken during a cold start period in the morning where the heating system was switched on at 6.40 am and turned off at 7.45 am, giving a total 'on' duration of 65 minutes. However, within this 65-minute period, there were seven start-ups and the actual time the boiler fired for was 53 minutes. The initial heating period lasted for 36 minutes. The first ten minutes of these are classified as 'cold start', with the remaining 26 minutes deemed to be steady state output. Thereafter, the boiler ran intermittently. During this third phase, the average 'on' time for each cycle was three minutes while the average 'off' time was two minutes. By examining these traces over an extended period of time, a definitive duty cycle for a typical installation can be obtained.

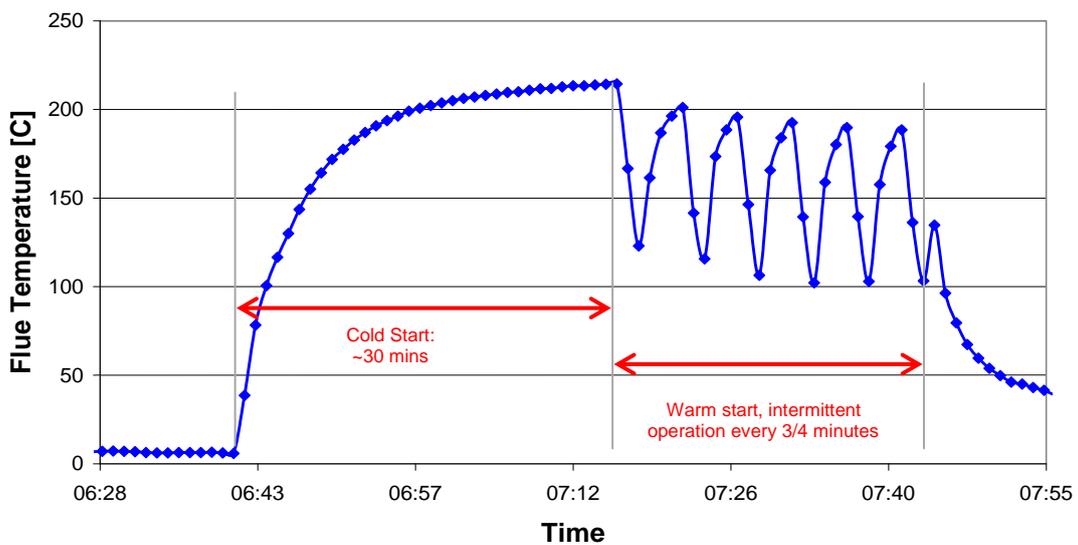


Figure 5.1. Typical time-temperature trace for a domestic oil-fired boiler

A comprehensive list of results from the duty cycle analysis of the six domestic oil-fired boilers is given in Morrin [133]. All six boilers followed the three phases, as described previously, and the general trend seen in [Figure 5.1](#) was replicated in all samples. The total running time for each boiler depends on a great many factors, such as occupancy rate, user perceptions, ambient temperature, insulation levels within the dwelling, etc. Any attempt to quantify the effect of these factors would have been outside the scope of this experiment, so the total run time is not normalised and, hence, should not be compared directly. Similarly, it is not possible to objectively compare the total number of cold or warm starts, as this is intrinsically linked to the total running time. What are directly comparable between installations are the average on and off durations found during start-up, average steady state duration, as well as the ratio of each of the three phases to the total running time found during each measurement period. The average on and off times during the warm start cycling phase were 3:54 minutes and 4:20 minutes respectively. The cold start period accounted for 13.4% of total running time on average. 24.1% of total run time was during the steady state phase, while the warm start operating cycle accounted for 62.4% of total running time on average.

5.2.2 Gas boiler duty cycle analysis

A gas-fired boiler typically follows the same firing cycle as an oil-fired boiler. In other words, the boiler goes through an initial start-up period, termed the cold start phase, after which it runs at steady state output, with only gradual changes in the system temperature until it reaches a set temperature. At this point, the boiler switches off and the system

begins to cool. When the temperature falls below a lower threshold, the boiler switches back on and fires at full output until it exceeds the upper temperature limit and the process repeats. This period is termed the 'warm start' phase. In a manner similar to the oil boiler analysis, it was possible to analyse the time-temperature plot for each of the gas boilers to determine a suitable duty cycle for each monitored installation.

Details of the results from the gas boiler analysis are given in Morrin [133]. The average on and off durations during this warm start phase were very repeatable across all of the installations with an average on duration of 3:40 minutes and an average off duration of 2:58 minutes. Two of them had a similar duty cycle to the oil boilers and, in their case, the cold start, steady state and warm start phases accounted for 16.0%, 31.1% and 52.9% of total running time respectively. However, in the case of boiler three and four, the duty cycle followed a different trend and spent very little time operating in the warm start mode. This is because these boilers had been recently installed in their respective houses, a new build in the case of boiler three and an extensive house renovation/upgrade in the case of boiler four. As a result, the boilers and the heating system had been correctly sized and, as a result, the warm start phase had a much smaller contribution to make to the overall running time for the boiler.

5.2.3 Wood-pellet boiler duty cycle analysis

Analysis of the wood-pellet boiler duty cycle analysis requires a little more detail than either the oil- or gas-fired boilers. The wood-pellet boiler is designed to modulate its output in an attempt to match the heat

demand of a building. In other words, it does not rely on the simple on-off control used in the boiler described previously. The main reason for this is that the ignition period for wood-pellets is very long when compared to an oil or gas boiler, approximately 5-10 minutes, as opposed to 30 seconds. Initially, a fixed quantity of pellets is fed into the combustion chamber and heated with an electrical element. A variable speed combustion fan delivers air to the pellets and once combustion has started, the boiler then goes through a flame stabilisation process with moderate inputs of fuel and air until the flame has been established. After that, the boiler operates at full capacity.

As the system temperature approaches the set point on the thermostat, the fuel feed rate and combustion air supply and resulting heat output are reduced. An example of a wood-pellet boiler duty cycle is shown in Figure 5.2. There are three distinct phases to each firing cycle, described as follows:

1. Start-up:

This is similar to the 'cold start' phase defined for oil- and gas-fired boilers. It is defined as the duration from when combustion is first initiated to the end of the flame stabilisation phase. As seen in the temperature trace, there is an initial increase in flue temperature, after which it dips as the fuel and air supply are reduced while the flame stabilises. The next phase is the full power steady state operation.

2. Full Power:

During this phase, the boiler runs at 'Power Level 5' (PL5), which is the maximum output it can achieve. During this time, the combustion fan speed and pellet flow rate are set to

their highest settings. During this period, there is a continuous gradual rise in flue gas temperature.

3. Modulation:

As the system temperature approaches the thermostat limit, the boiler begins to modulate its output, by reducing the fuel and air supply. In total, there are five discreet power levels (PL1 to PL5), each with their own fuel and air supply settings. As temperature increases further, the output drops to a lower power level in an attempt to match the heat demand. If, even after reducing to PL1, the heat output is greater than demand and the maximum temperature setting on the thermostat is exceeded, the boiler shuts down. It then tends to remain off for a relatively long period of time. The reasons for this long non-operational period is that the volume of water contained within the heating jacket in the boiler is large (65 litres) when compared to other heating systems, and thus has a large thermal inertia and acts as a buffer. In addition, many wood-pellet heating systems make use of a large buffer tank (1000+ litres) which acts as a thermal store, which can supply the building while the boiler is switched off.

For the purposes of analysing the duty cycle data, a number of assumptions are made and applied to all three monitored boilers. The 'cold start' period is defined as the time from the initial temperature rise, to the point where temperature begins to rise again after the stabilisation period, as highlighted in [Figure 5.2](#). This period must also occur after

a long shut-down period, where the system temperature would have cooled back down to ambient levels again. If the firing cycle begins only a short duration after a previous shut down, it is termed a 'warm start'. The second phase, the full power 'steady state' phase, is defined as the duration from the end of the stabilisation phase to the point where the flue temperature begins to decrease. This point is then taken as the start of the modulation phase and the point where the boiler shuts down is obvious, as

the flue temperature falls away very rapidly. As described earlier, there are four separate steps in the modulation phase, as the boiler reduces output step by step. Unfortunately, with the measurement frequency used (one measurement/minute), the resolution is not high enough to definitively know the precise duration at each power level. Instead, it is assumed that the time spent at each power level is equal and is found by dividing the total modulation time by four.

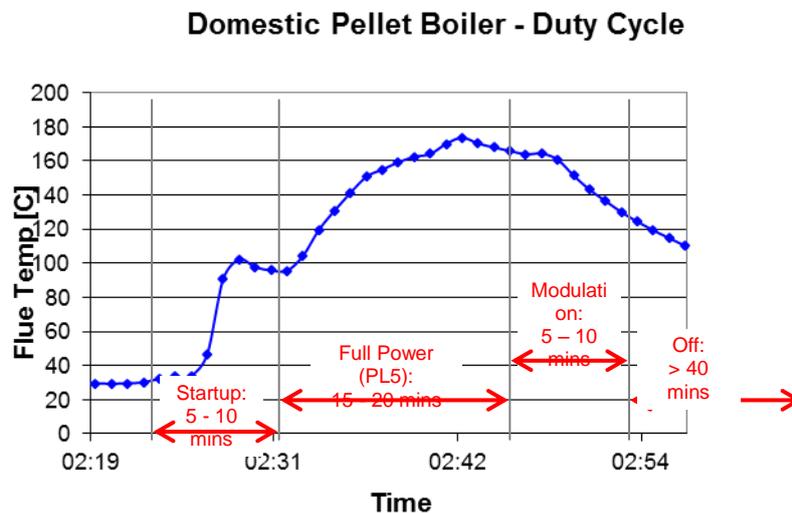


Figure 5.2. Sample wood-pellet boiler duty cycle

A summary of relevant results for each of the three boilers is given in Morrin [133]. The main points of interest that can be gleaned from the table are the relative contribution of the steady state full power output and the start-ups have on the overall running time. The PL5 contribution was similar for all three installations, accounting for up to three-quarters of total time. The modulation periods played a small role, accounting for less than 10% of total time. There is some variation in the cold and warm start-up times. Boilers one and two spent short periods of

time in the 'cold start' mode, and in the case of boiler two, spent a lot of time in warm start mode. This tells us boiler two tended to shut down, remain off for a relatively short period of time (<40-50 minutes) and then re-start. In contrast, boiler one spent much longer at PL5 and once it shut down, it stayed off for long periods, explaining the low contribution of warm starts to the overall running time for boiler one. Finally, boiler three exhibited similar characteristics to boiler one, in that it did not enter warm start mode very often, but rather stayed at PL5 for extended periods of

time. It spent longer at cold start than either boilers one or two, despite have less cold starts per day on average, and this can be explained by the lower boiler rating of 20kW, which meant it took longer for the boiler to heat the house from cold. This data was then used in conjunction with the emission factors measured in the laboratory and accurate weighted-average emission factors could be derived.

5.3 Particulate matter emission factor measurements

In this section, details of all particle matter measurements carried out in the lab will be described in detail. First, details of measurements performed on the wood-pellet boiler will be described, before moving on to results from the tests on the oil and gas boilers.

5.3.1 PM from wood-pellet boiler

In 2006, Sustainable Energy Ireland launched the Greener Homes Scheme [121] which made €27 million available for domestic renewable heating technologies, such as heat pumps, solar panels, wood-pellet stoves and boilers. Grant aid of up to €4,200 was available for households when purchasing a wood-pellet boiler heating system. Prior to this scheme, there was no market for wood-pellet boilers in the country. By the end of 2010, over 5,700 homes had availed of the grant scheme and had installed wood-pellet boilers, and it is estimated that this will increase in the coming years [14]. In Eastern Europe and Scandinavia, however, wood-pellet boilers have been in use for many years and are a mature technology. A number of advanced wood-pellet boilers, such as the KWB

easyfire, [122] and the ETA PE range [123], use sophisticated control systems, such as exhaust gas oxygen sensors in conjunction with fuel and/or air staging, to improve efficiency while reducing emission levels.

The pellet boiler chosen for use in the experimental test facility was a Gerkros Woodpecker boiler, with a nominal rating of 20kW. This model was chosen as it was deemed to be a typical example of the type of wood-pellet boiler that had been purchased through the Greener Homes scheme. In addition, it had been designed and manufactured in Ireland and was developed with the Irish climate and heating demand in mind. Heating system requirements in Eastern countries differ greatly from those encountered in Ireland. In colder climates, the heating demand is much higher and boilers are designed to run uninterrupted for long periods of time, providing a constant supply of heat. Our winters tend to be quite mild and, as a result, heating boilers run intermittently. The Woodpecker boiler used an automatic pellet ignition system and was able to modulate its heat output in response to changing heat demands, and it was decided that this particular model was the most representative of a typical Irish installation.

A series of tests were performed on the pellet boiler to assess both efficiency and pollutant emission levels across all operating conditions. The wood-pellets used are commercially available, are manufactured from Sitka spruce (*Picea sitchensis*) and conform to the CEN / TS 14961 standard. [29] The pellets have a diameter of 6mm and a lower heating value of 17.292 MJ kg⁻¹ [124]. The elemental composition values were taken from the Phyllis database which

has been developed by the Energy Research Centre of the Netherlands [125].

The properties of the pellets used are presented in [Table 5.1](#).

Table 5.1. Wood-pellet characteristics

Pellet Composition (dry wt %)	
<i>Carbon</i>	50.0
<i>Hydrogen</i>	5.9
<i>Oxygen</i>	43.9
<i>Nitrogen</i>	0.2
Moisture (%)	8.0
Lower Heating Value (MJ/kg)	17.29
Average Dimensions	
<i>Diameter (mm)</i>	6.0
<i>Length (mm)</i>	10.7
Average mass (g)	0.392

During each test, the gaseous composition of the flue gas was constantly monitored using the Testo 350XL gas analyser, and the particulate matter was sampled for a set period of time with the low pressure impactor. A comprehensive series of tests were performed on the wood-pellet boiler to examine its performance and emission characteristics across all operating modes. As described previously, the pellet boiler had a number of different operating modes, referred to as 'power levels' (PLs). The boiler output was allowed to settle so that all temperatures remained stable and output matched demand, a condition described in this study as 'steady state'. A number of tests were performed at each of these power levels with the initial emphasis on PL5. Power level 5 represents the optimal

combustion conditions that the boiler was designed for, and a total of 11 tests were performed in this mode. As the results from the wood-pellet boiler duty cycle analysis show, this is the main operating mode for the boiler and accounts for up to 73% of total boiler running time. The other four power levels were then examined in turn, following the same procedure as for PL5. The boiler was allowed to settle at steady state output at each power level and then sampling could begin. Finally, the concentration of PM was measured during the start-up process. The start-up procedure in the pellet boiler takes a reasonably long period of time (~15 minutes), as the flame first has to be initiated, and then allowed to stabilise. The PM concentration during this period was too high to sample in a single test, so instead the

period was divided into seven distinct phases, determined by the flue temperature, and PM concentration in each phase measured. In this way, both the overall

concentration, and the change in concentration over the whole start-up period could be measured. A summary of results is given in [Table 5.2](#).

Table 5.2. Summary of PM test results for wood-pellet boiler at each operating mode

Operating Condition	Fuel Energy Input [kW]	No. of Tests	Excess Air Ratio λ	Flue Temp [C]	PM [g/GJ]	Uncertainty	Repeatability
PL5	24.2	11	1.55	145.3	14.6	20.3%	40.4%
PL4	19.1	3	1.60	122.0	21.5	25.5%	11.2%
PL3	16.8	2	1.57	110.0	24.9	40.3%	13.0%
PL2	13.9	2	1.79	105.0	29.0	54.2%	15.4%
PL1	5.4	1	2.64	96.1	51.4	76.2%	-
Startup	varies	1	11.40	19.9	68.2	82.3%	-
		1	2.87	37.4	30.8	79.6%	-
		1	1.25	74.9	18.0	72.8%	-
		1	1.15	88.0	24.4	71.8%	-
		1	1.45	94.7	15.9	77.7%	-
		1	1.42	104.6	14.5	70.7%	-
		1	1.66	118.0	10.6	28.7%	-

Where multiple tests were performed at a particular operating mode, the repeatability of the tests is presented as ± 2 standard deviations (95% confidence interval) in [Table 5.2](#). As the results show, the PM emission factor was strongly dependent on the boiler load. As PL5 represents the optimal combustion conditions in the burner, the level of PM emitted in this state is relatively low at 14.6 g/GJ \pm 20.3%. This value is in general agreement with emission factors

reported elsewhere [[108](#), [126](#)]. As the load decreased, however, the emission factor increased quite dramatically, i.e., – a three-fold increase in the case of power level 1. There was an even higher PM concentration during the start-up phase, with a peak of 68.2 g/GJ \pm 82.3%. The high uncertainty level in this case is due to the large uncertainty in the quantity of fuel consumed. A fixed amount of pellets was delivered to the combustion chamber during that

particular phase but there was no way of accurately quantifying the precise amount that had been consumed by the end of the

testing period. A graph showing the variation in PM emission factor for each of the operating modes is shown in [Figure 5.3](#).

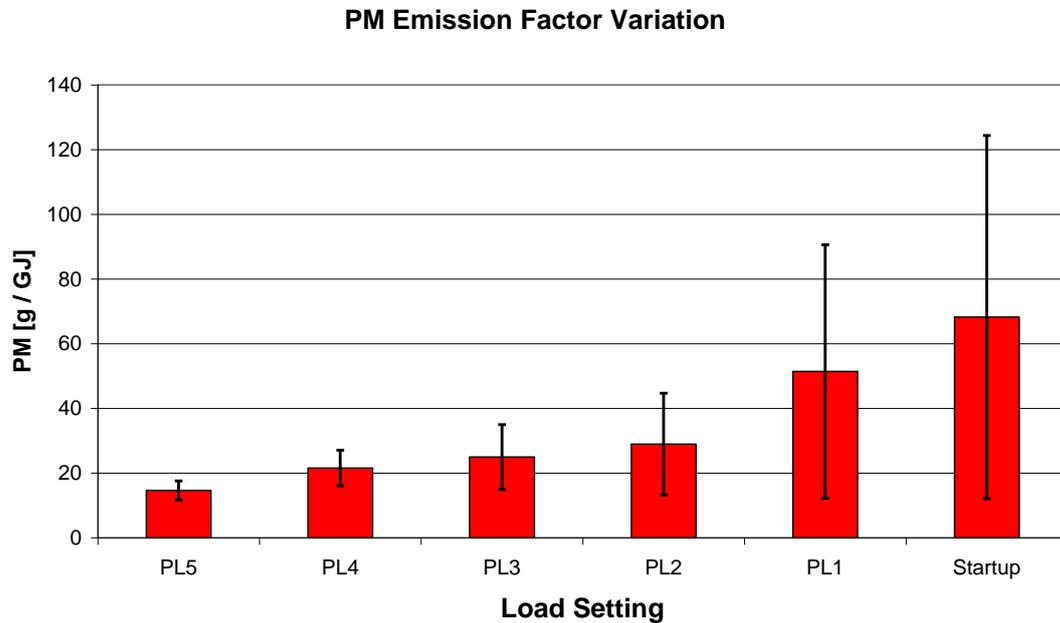


Figure 5.3. Measured PM emission factors and associated uncertainty limits from combustion of wood-pellets

[Figure 5.4](#) shows the evolution of the PM emission factor over the course of the start-up period. From an initial peak of 68.2 g/GJ, the emission factor falls quite rapidly as the flue temperature, and, hence, combustion temperature begins to rise. When the start-up phase eventually finishes the boiler then runs at steady state PL5 for an extended period of time until the modulation phase

begins. The PM concentration measured in the final start-up test was 10.6 g/GJ \pm 28.7%, somewhat lower than the average value found in the eleven PL5 tests that had been performed but still within the latter's uncertainty band. The lower value in this case can simply be put down to normal variations between tests.

Variation of PM emission factor and flue temperature during startup

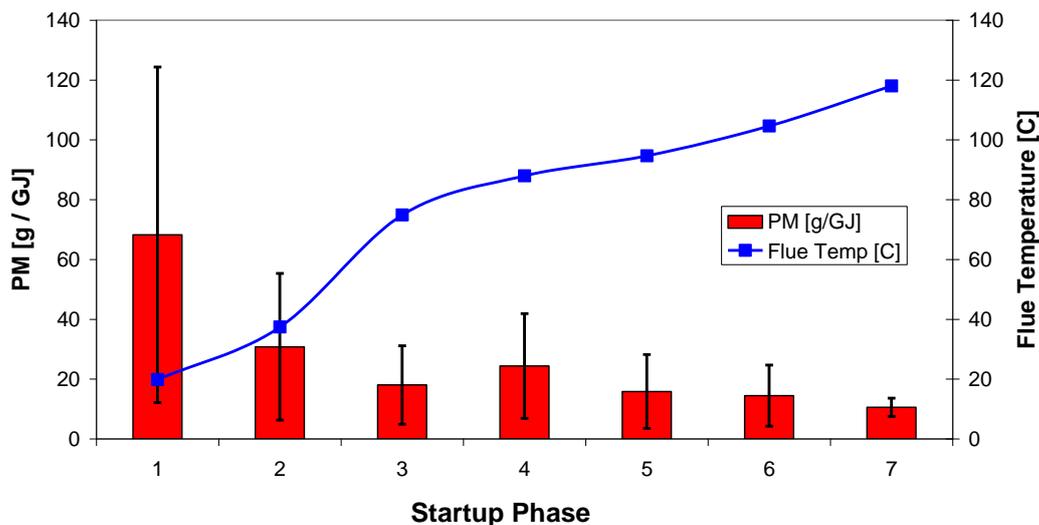


Figure 5.4. Variation of PM emission factor and flue gas temperature during the start-up phase

Finally, the other very important piece of information that the impactor gives is the mass size distribution. The size distribution was found to be very repeatable for all tests and consisted of a large peak between approximately 90nm and 400nm and then minor fractions of coarse particles. The average mass size distributions for each of the five power levels, as well as during start-up, are shown in [Figure 5.5](#). As the particle size is shown on a logarithmic scale, the

spacing between successive stage cut points varies. If the graph was presented with size interval between stages on the X-axis and particle mass on the Y-axis, the results would be distorted. To overcome this, the results are normalised by dividing the mass of collected particles by the width of the interval between stages. The impactor measures the mass of collected particles relative to the total mass of flue gas sampled, and the units are in mg/Nm³.

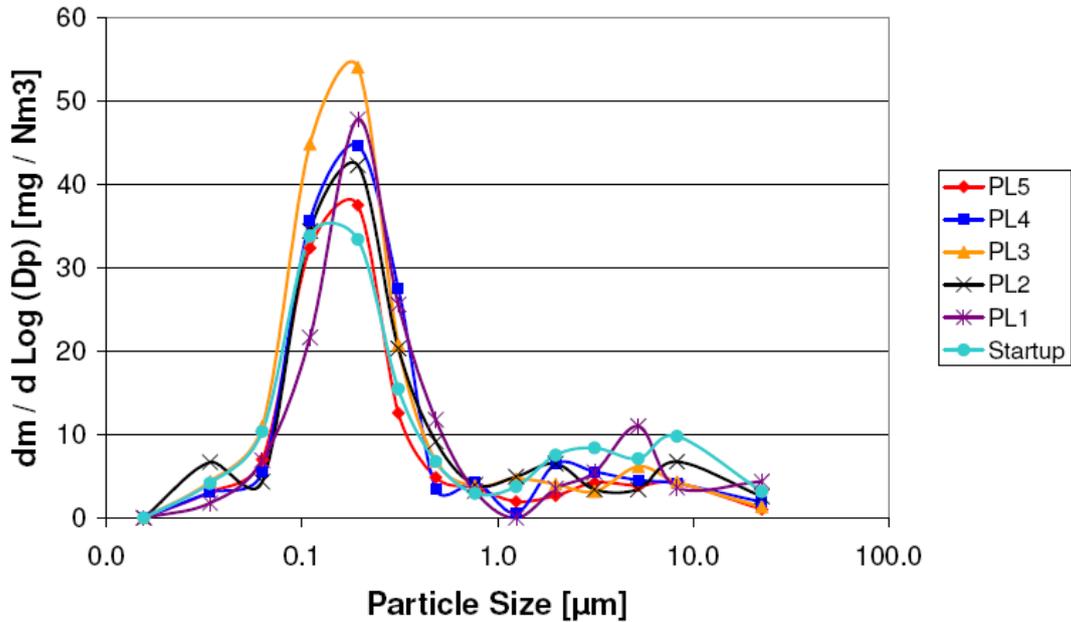


Figure 5.5. Average mass size distribution for all operating conditions

5.3.2 PM from oil boiler

The oil boiler that was chosen for testing was a Firebird Enviromax C26 condensing boiler, fired with kerosene fuel. The boiler had a nominal thermal input rating of 26kW. As outlined in [Section 2.4](#), the fuel is supplied under constant pressure to a nozzle. The oil is accelerated as it passes through the nozzle causing it to atomise into a fine mist. Combustion is initiated by sparking electrodes and, once started, continues to burn as long as there is a continuous supply of atomised fuel.

A comprehensive series of test was performed on the boiler to examine its performance and emission characteristics for all possible operating modes. The three main operating modes investigated were:

1. Steady state operation: For these tests, the boiler was brought up to operating temperature and stabilised by matching the cooling rate to the heat input, so that all

temperatures stayed constant for the test duration.

2. Cold start tests: These tests sought to replicate the operation of a boiler when starting up after a long non-operational period. The hypothesis was that, with a cold combustion chamber and water heating jacket at ambient temperature, the flame temperature would be lower than it would be at optimal conditions and emissions would increase.
3. Warm start tests: Similar to the cold start tests, these sought to measure the emission concentration produced during intermittent operation of the boiler. As described earlier, a typical Irish installation boiler operation is characterised by frequent on-off cycling to maintain the set point temperature for the heating system.

For each of these modes, three samples were taken using the DLPI, and a further three samples taken with a filter system,

designed as an additional measure for particulate matter to be used in conjunction with the impactor. The composition of the flue gas was measured continuously during all tests using the Testo 350 gas analyser. The main operating parameter that could be modified during the tests was the air:fuel ratio. The burner had a damper at the air inlet, which essentially was a butterfly valve. By closing the valve, less air could be supplied to the combustion chamber, enriching the mixture while opening the valve supplied more air. Over the course of a test, the AFR could vary quite dramatically from start-up to steady state mode, so it was necessary to monitor the flue gas composition and adjust the air damper to maintain as constant an AFR as possible.

The initial set of tests performed on the oil boiler was done using the DLPI, while the boiler was running in steady state operation. The results from the steady state tests are

presented in [Table 5.3](#). The sample time was governed by a couple of factors. For the DLPI, the onset of particle bounce limited the sample time. This optimum sampling time was found through a process of trial and error. Ideally, the sample time should be as long as possible so as to give a measurable increase in mass for each collection substrate, yet short enough to avoid overloading of the substrate and consequent particle bounce. For the filter, the sample time was decided either by the filter pores beginning to clog or where concentrations were low enough for this not to be an issue, a sample time was chosen that yielded an increase in filter mass that was easily measurable. As seen in the results table, the measured emission factor was higher when using the filter sampling method as opposed to the DLPI. This can be attributed to losses in the impactor itself, as well as the different sampling conditions.

Table 5.3. Summary of PM results for steady state sampling of oil boiler

Sampling Method		Test Number			Average	Average Uncertainty
		1	2	3		
DLPI	PM [g / GJ]	0.21	0.21	0.28	0.23	19.6%
	λ	1.28	1.28	1.28	1.28	0.8%
	Sample Time (mins)	120	120	113		
Filter	PM [g / GJ]	0.38	0.39	0.40	0.39	12.9%
	λ	1.31	1.31	1.33	1.32	0.8%
	Sample Time (mins)	120	150	150		

The next set of tests carried out on the oil boiler looked at its performance during cold start-ups, and the results are presented in [Table 5.4](#). The cold start-up period was defined as the first ten minutes under constant load. After this initial ten minute period, the flue gas temperature and boiler output started to level off and find a steady state. The cooling fan speed was set to a constant value of 675 rpm for all cold start tests, so that that the simulated cold start load remained unchanged between tests. After each ten minute sample period had been completed, both the boiler and the sampling pump were switched off. The cooling fan was then set to its maximum

speed to cool the system temperature back down to ambient levels again. Even though the boiler had stopped firing, the water circulation pump remained running facilitating the continued heat exchange. Once the water in the circulation system had reached ambient temperature, the boiler and sampling pump were switched back on again and another ten minutes of cold start-up emissions were sampled. This was repeated until a measurable mass of PM had been collected. The results from these cold start tests exhibited similar trends to the steady state tests, in that the filter sample gave a higher emission factor than the DLPI.

Table 5.4. Summary of PM results for cold start sampling of oil boiler

Sampling Method		Test Number			Average	Average Uncertainty
		1	2	3		
DLPI	PM [g / GJ]	0.29	0.30	0.25	0.28	21.1%
	λ	1.27	1.30	1.31	1.29	0.8%
	Sample Time (mins)	100	120	110		
Filter	PM [g / GJ]	0.35	0.30	0.30	0.32	14.9%
	λ	1.31	1.52	1.28	1.37	0.7%
	Sample Time (mins)	150	140	150		

Next was an investigation into the behaviour of the boiler during warm start transient operating mode. As outlined previously, in a typical installation, the operation of the boiler is characterised by frequent on-off operations and a high number of start-ups,

the aim of which is to keep the temperature of the heating system close to its set point. The warm start tests were achieved by controlling relevant parameters, such as cooling fan speed and mixing valve position on the experimental facility, so as to induce a

duty cycle with similar on-off times as measured in a typical domestic installation using the data loggers. Once the duty cycle was set up, it was then repeated over a long duration of time, and the particulate emissions sampled while the boiler was running. When the boiler shut down, the sampling pump was switched off within a

matter of seconds and then switched back on just before the boiler started back up again. In this way, the sampling system was only running for the times the boiler was in operation, and not while it was turned off. The results for the warm start tests, as measured using the DLPI and filter sampling system, are given in [Table 5.5](#).

Table 5.5. Summary of PM results for warm start sampling of oil boiler

Sampling Method		Test Number			Average	Average Uncertainty
		1	2	3		
DLPI	PM [g / GJ]	0.35	0.37	0.31	0.34	14.9%
	λ	1.27	1.36	1.32	1.32	0.8%
	Sample Time (mins)	120	121	123		
Filter	PM [g / GJ]	0.27	0.28	0.29	0.28	15.1%
	λ	1.37	1.41	1.54	1.44	0.6%
	Sample Time (mins)	147	144	155		

Once these initial tests examining PM concentration at three different operating modes had been carried out, the effect of the relative air:fuel ratio λ on particulate emissions was investigated. Seven separate samples were taken using the filter sampling

system and, in each test, the AFR was changed and the boiler was allowed to settle into steady state operation with that particular AFR. The results of these tests are shown in [Table 5.6](#).

Table 5.6. Effect of relative air:fuel ratio λ on PM emission factor

Test	PM [g/GJ]	λ
1	2.40	1.15
2	0.69	1.24
3	0.34	1.43
4	0.30	1.52
5	0.23	1.64
6	0.25	1.70
7	0.08	1.83
Average	0.61	1.51
Uncertainty	9.9%	0.9%

As [Figure 5.6](#) shows, the PM emission factor gradually increased as the AFR decreased before rising very sharply once the level of excess air fell below approximately 25% ($\lambda = 1.25$). The trend in this graph is very similar to the one shown for CO in [Figure 2.6](#). In it,

the theoretical variation of CO against AFR was plotted and, in that case, when the excess air level dropped below approximately 5%, the CO concentration increased very rapidly, in much the same manner as for PM shown in [Figure 5.6](#).

Variation of PM emission factor Vs Relative AFR

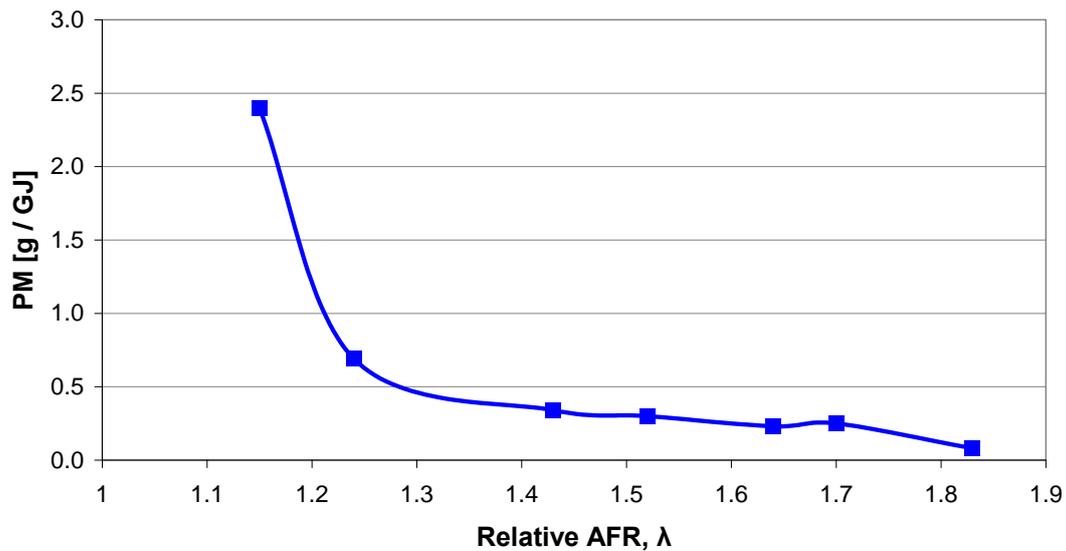


Figure 5.6. Plot of particulate matter emission factor against the relative air:fuel ratio λ , showing the exponential correlation between them

While the experimental facility used in this study could simulate all types of operating loads found in a typical installation, the one aspect of boiler operation that it could not replicate was that of a boiler with a worn nozzle. As outlined in [Section 2.4](#), clean and efficient burning in an oil-fired boiler depends mainly on the atomisation of the fuel. This is achieved by accelerating the fuel through a fine nozzle, whose diameter is measured in fractions of a millimetre, causing it to atomise into a fine mist. Over time, the dimensions of the nozzle in the burner can change due to wear, and adversely affect the degree to which the fuel can be atomised. It may take a number of years for this process to occur, something which was not practical to test in the lab.

However, burner nozzles are a service item and it is recommended that they are replaced at each annual service. In practice,

this rarely happens, so in order to examine what effect, if any, a worn nozzle had on the particulate matter emission factor, two worn nozzles that had been in use for over two years each were acquired and fitted to the existing boiler in the laboratory. These nozzles had been deemed past their useful life by a service technician during a routine service on a boiler installed in a domestic house. For each nozzle, the boiler was subjected to two steady state filter sample tests, using the same process that was used for all other steady state tests and the particulate matter emission factor was calculated. It was found that for the two old nozzles tested, no discernible difference in the PM emission factor could be found when compared to the brand new nozzle used for all other PM tests. The results of these tests using worn nozzles are presented in [Table 5.7](#).

Table 5.7. Results of PM emission factor tests carried out using two old nozzles that had been replaced on a boiler installed in a domestic house

	Old Nozzle 1		Old Nozzle 2	
	PM [g/GJ]	λ	PM [g/GJ]	λ
Test 1	0.33	1.35	0.28	1.34
Test 2	0.33	1.33	0.30	1.33
Average	0.33	1.34	0.29	1.34
Uncertainty	12.8%	0.8%	15.1%	0.8%

A final set of tests were performed on the oil boiler to investigate whether flue gas condensation had any effect on particulate matter concentration. The steady state tests were repeated, and PM concentration sampled using the filter system, but this time

the boiler was 'forced' to condense over the full duration of the test. By setting the cooling fan to its maximum speed, the cooling rate was considerably higher than the rate of heat output from the boiler, and this kept the flue temperature suppressed to 60-65°C for the

duration of the test. The dew point temperature for these tests was 44.3°C, considerably lower than the temperature of the bulk flue gas. However, it is thought that the cooler temperature found at the heat exchanger walls caused the temperature to drop below the dew point and allow water condensation to occur. Two tests with the same operating conditions were carried out in this mode and the results were conclusive. The average PM emission factor was 0.340 g/GJ ± 14.5%, while previous steady state

tests, using the same experimental parameters but without water vapour condensation, gave an emission factor of 0.387 g/GJ ± 12.9%. The values from both sets of tests are very similar, and lie within each other's uncertainty limits. Running the boiler in condensing mode was found to have no significant impact on the concentration of particulate matter emitted. The results of these two tests are given in [Table 5.8](#).

Table 5.8. Results from the 'forced' condensing tests performed on the oil boiler to investigate whether particulate concentration was influenced by water vapour condensation

	PM [g / GJ]	λ	Flue Temperature
Test 1	0.32	1.34	66.2
Test 2	0.36	1.34	61.8
Average	0.34	1.34	64.0
Uncertainty	14.5%	0.8%	1.7%

5.3.3 PM from gas boiler

The gas boiler that was used in the laboratory was a Baxi Solo 18HE condensing boiler. The boiler had a thermal input rating of 20.18kW and had a maximum rated output of 19.3kW. Gas is supplied to a gas-air mixing valve where air is entrained in the flow and mixed with the gas prior to entering the burner. The mixing valve settings are adjusted by the manufacturer to mix a fixed quantity of air with the fuel and are non-adjustable. The gas-air mixture then flows to a manifold where combustion is initiated by a pilot flame.

A number of tests were performed on the gas boiler to measure both its efficiency and emissions for all operating modes. As was the case for the oil boiler, three operating modes were examined: steady state operation, cold and warm start-ups. For the steady state tests, the cooling rate was set at a constant value and the system was allowed to settle at a steady output. Once the output had stabilised, sampling could begin. The cold start period was once again taken as the first ten minutes after starting from cold. The boiler and sampling equipment ran for these ten minutes and then shut down, and the cooling fan set to

maximum speed to cool the system back down to ambient temperature. This process was repeated over 30 times so that a measurable quantity of PM was collected using the filter sampling system. The DLPI was not used on the gas boiler, as the particle concentration was so low that it was outside the measuring range of the impactor, and no measurable amount of PM could be gathered in a reasonable time frame. The warm start tests sought to measure emission levels during the intermittent operation, as determined by the duty cycle measurements.

The scope for performing other tests on the gas boiler was more limited than for the oil boiler tests as it was not possible to adjust the air:fuel ratio. The gas-air mixing valve settings were non-adjustable, so an investigation into the effect of the AFR on emission levels could not be carried out. No investigation into wear and aging of gas

burners was performed. It was found that the emissions characteristics of the gas boiler differed significantly from the previous boilers, as the quantity of particulate matter produced was orders of magnitude smaller and, hence, almost negligible. In the following section, PM emissions at each operating state are presented.

The first tests that were carried out sought to measure the levels of PM produced during steady state operation of the gas boiler. The tests were repeated four times, maintaining test conditions as similar as possible to allow for comparable results, and the average value taken to be the steady state emission factor. The sample time was governed by the need to collect a measurable mass of PM on the filter paper and tests lasted up to eight hours. The steady state test results are given in [Table 5.9](#).

Table 5.9. Results from steady state tests performed on the gas-fired boiler

Test	PM [g / GJ]	λ	Efficiency [%]	Flue Temperature [C]	Sample Time [mins]
1	0.03	1.21	92.9%	60.9	197
2	0.04	1.22	90.7%	60.2	300
3	0.02	1.24	92.0%	61.3	394
4	0.03	1.23	92.5%	61.7	480
Average	0.03	1.23	92.0%	61.0	342.75
Uncertainty	23.2%	1.1%	8.0%	1.8%	

As can be seen in the table of results, the test conditions (relative air:fuel ratio λ , flue temperature) were very repeatable. As this is

a condensing boiler, the flue gas temperature was very low when compared to older boilers that did not allow the water

vapour in the flue gas to condense. The dew point temperature for combustion of gas was calculated as 55°C, so while the amount of condensation that actually occurred during these steady state tests was very small, there was an increase in efficiency, as less heat was being rejected to the atmosphere. The efficiency values listed here are based on the higher heating value, which takes into account the latent heat of vapourisation of water that gets released during the condensation process.

The final pair of tests performed on the gas boiler examined its performance during both cold and warm start-ups. The cold start test period was defined as the first ten minutes of operation, and the test was repeated 31

times so as to collect a sufficient quantity of PM. The measured emission factor for cold start-ups was 0.02 g/GJ. The quantity of PM produced under these conditions was negligible and the tiny emission factor reflects this. For the warm start tests, a suitable duty cycle was derived from the duty cycle measurement programme and replicated in the laboratory. The measured emission factor warm start-ups was 0.04 g/GJ. Again, PM levels were practically negligible. In both tests, the temperature of the flue gas was below the dew point temperature of 55°C and, hence, large quantities of condensate were generated. The results of these two tests are shown in [Table 5.10](#).

Table 5.10. Results from cold and warm start transient tests performed on the gas boiler in the laboratory

Test	PM [g / GJ]	λ	Efficiency [%]	Flue Temperature [C]	Sample Time [mins]
Cold start	0.02	1.26	82.4%	45.0	310
Uncertainty	27.9%	1.1%	8.0%	2.5%	
Warm start	0.04	1.30	51.7%	50.4	297
Uncertainty	22.4%	1.1%	12.0%	2.2%	

5.3.4 Summary of PM measurements

A comprehensive series of tests was performed on all three boilers in the

laboratory and a summary of the measured emission factors is given in [Table 5.11](#) and [Table 5.12](#) below.

Table 5.11. Wood-pellet emission factors to be used for compiling emission inventories

Operating Mode	PM [g/GJ]
PL5	14.62
PL4	21.55
PL3	24.95
PL2	28.97
PL1	51.41
Start-up	35.37

Table 5.12. Results from steady state tests performed on the gas-fired boiler

	Oil	Gas
Test Condition	PM [g / GJ]	PM [g / GJ]
Steady State	0.39 ± 12.9%	0.03 ± 23.2%
Cold Start	0.32 ± 14.9%	0.02 ± 27.9%
Warm Start	0.28 ± 15.1%	0.04 ± 22.4%

5.4 NO_x emission factor measurements

Similar to the results of the particulate matter measurements, this section describes the results of the NO_x emission factor measurements. These measurements were taken from the same tests as the PM samples and both tests were done at the same time. First, the NO_x emissions from the wood-pellet boiler will be described, before moving on to the results from the oil and gas boilers.

5.4.1 NO_x emissions from wood-pellet boiler

As described in Chapter 3, the composition of the flue gas was monitored during all tests. Similar to the measurement of particulate matter emission factors, the effect of boiler load and start-up on the concentration of NO_x produced was investigated. The results presented here correspond to the same tests used to measure PM concentration, as outlined in Section 5.3. The results of the NO_x measurements taken during the steady state and start-up modes are shown in [Table 5.13](#).

Table 5.13. Summary of NO_x test results on wood-pellet boiler

Operating Condition	Fuel Energy Input [kW]	No. of Tests	Relative air:fuel Ratio λ	Flue Temp [C]	NO _x [g/GJ]	Uncertainty	Repeatability
PL5	24.2	11	1.55	145.3	57.1	15.4%	28.4%
PL4	19.1	3	1.60	122.0	48.0	20.8%	3.3%
PL3	16.8	2	1.57	110.0	44.9	31.2%	0.5%
PL2	13.9	2	1.80	105.0	43.0	42.3%	0.5%
PL1	5.4	1	2.64	96.1	45.4	62.6%	-
Start-up	varies	1	11.40	19.9	25.5	145.0%	-
		1	2.87	37.4	25.1	67.0%	-
		1	1.25	74.9	49.2	56.0%	-
		1	1.15	88.0	47.8	56.0%	-
		1	1.45	94.7	51.6	56.1%	-
		1	1.42	104.6	51.5	56.1%	-
		1	1.66	118.0	49.8	17.9%	-

In contrast to the PM tests, the concentration of NO_x emissions was not strongly dependent on boiler load. The NO_x emission factor was fairly constant across the five power levels, the only significant deviation occurring during PL5 operation. Where multiple tests were performed, the measurements were all repeatable, as evidenced by ± 2 standard deviations about the mean (95% confidence interval). During start-up, the emission factor started out at a

low level of 25.5 g/GJ but quickly rose to match the concentration found during steady state operation at each of the power levels. [Figure 5.7](#) shows graphically the effect of the boiler load setting on the NO_x emission factor, along with their associated uncertainty bands. It is clear that all five emission factors fit within each other uncertainty limits, reinforcing the fact that NO_x concentration is not strongly dependent on load.

NO_x Emission Factor Variation

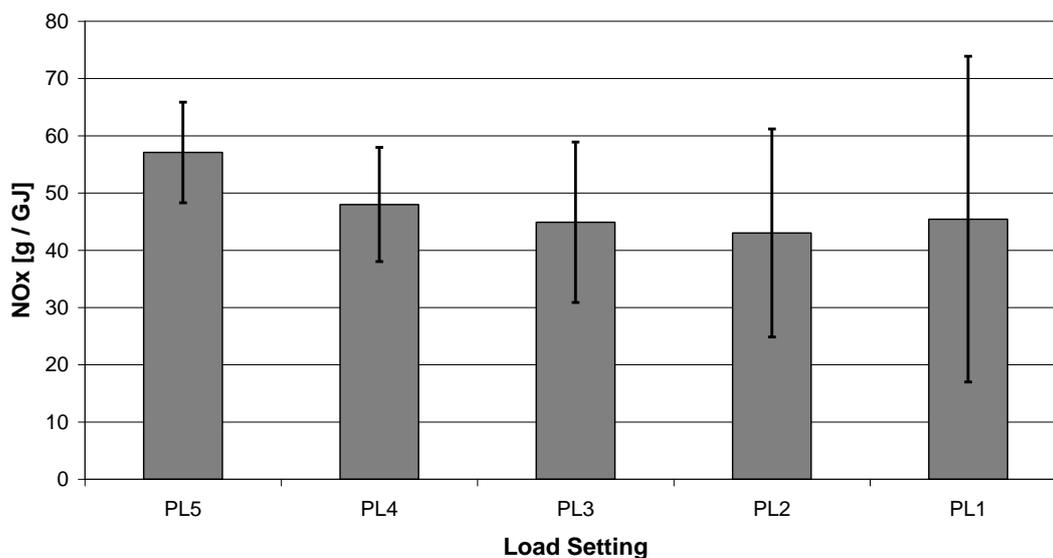


Figure 5.7. Effect of boiler load on NO_x emission factor

Figure 5.8, meanwhile, plots the change in NO_x emission factor and flue temperature over the course of the start-up phase. Once again, there is little variation once the initial heat-up phase is completed and the boiler settles out at approximately the same value as that measured during steady state

operation at each of the power levels. The uncertainty bands on the first six of these measurements are particularly large, as there was no way to accurately determine the mass of pellets consumed during a particular test.

Variation of NO_x emission factor and flue temperature during startup

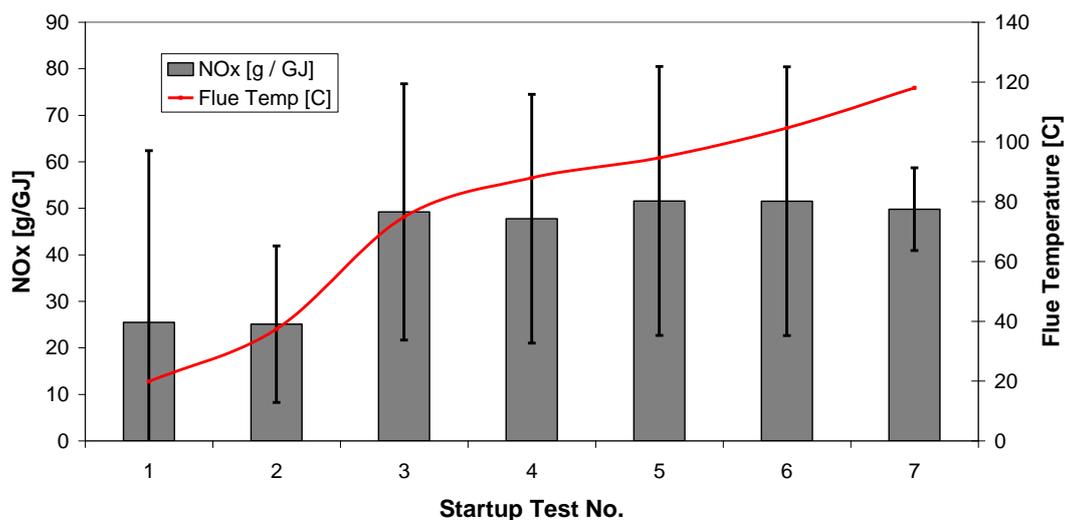


Figure 5.8. Variation in NO_x emission factor during the start-up phase

5.4.2 NO_x emissions from oil boiler

As described in the introduction to this section (Section 5.4), the gaseous composition of the flue gases were monitored continuously during all tests. In this way, both PM and NO_x could be measured simultaneously. As outlined in

Section 5.3.2, three main operating modes were examined: steady state operation, cold and warm start-ups. For each operating mode, six tests were performed, and test conditions were kept as repeatable as possible for each. The results for the steady state tests are given in [Table 5.14](#).

Table 5.14. Measured NO_x emission factors from steady state testing of oil-fired boiler

Test Number	NO _x [g/GJ]	λ	Boiler Efficiency η (%)	Flue Temperature [C]
1	42.95	1.28	96.7%	87.7
2	41.53	1.28	96.8%	86.6
3	42.78	1.28	96.0%	87.0
4	42.78	1.31	96.6%	84.6
5	42.87	1.31	97.4%	84.6
6	43.98	1.33	96.4%	84.4
Average	42.82	1.30	96.7%	85.8
Uncertainty	11.3%	0.8%	7.4%	1.3%

The results for the six steady state tests were very repeatable, and there was very little variation in the measured emission factors, with the average being 42.82 g/GJ. The air:fuel ratio, flue gas temperature and boiler efficiency values (based on lower calorific value) were all very similar for each of the six tests. The average efficiency for the six tests was just under 97%, indicating the very high efficiency levels achieved in modern heating boilers.

The next set of tests that were performed on the oil boiler examined the effect of cold start-ups on the NO_x emission factor. As

already described, the cold start period was defined as the first ten minutes under constant load. After this initial period, the flue gas temperature and boiler output began to level off and find a steady state value. The cooling fan speed was set to a constant value for all cold start tests, ensuring that the cooling rate was constant. After each ten minute sample, the boiler was shut down and the cooling fan set to its maximum speed to bring the system temperature back down to ambient. This process was then repeated 10-15 times. The results of the cold start-up tests are presented in [Table 5.15](#).

Table 5.15. Measured NO_x emission factors, and other parameters from the oil boiler cold start tests

Test Number	NO _x [g/GJ]	λ	Boiler Efficiency η (%)	Flue Temperature [C]
1	38.31	1.27	60.6%	46.8
2	39.96	1.30	59.9%	48.8
3	46.11	1.31	58.7%	48.8
4	42.62	1.31	60.1%	47.0
5	54.07	1.52	59.5%	48.8
6	47.56	1.28	61.1%	44.5
Average	44.77	1.33	60.0%	47.5
Uncertainty	11.1%	0.8%	8.0%	2.4%

The average emission factor for NO_x during the cold start-up period was found to be 44.77 g/GJ. As mentioned in the introduction to this section, the main variable for each test was the AFR, which was controlled by the position of a damper at the air inlet to the burner. The aim of each test was to keep this as constant as possible, so as to ensure constant and comparable conditions for each test. For the most part, this was achieved but in the case of test 5, additional air was supplied which had an impact on the measured emission factor.

The third operating mode that was tested was the warm start condition. As described

in Section 5.3.2, duty cycles that had been measured in actual domestic installations were replicated in the laboratory by controlling the relevant parameters, such as the mixing valve position and cooling fan speed. Once an appropriate duty cycle had been achieved, the sampling process was repeated several times until the overall sample time exceeded two hours. The results presented below relate to the duration the boiler was firing for, and flue gas readings taken while the boiler was shut down are not included. The results for the warm start tests are given in [Table 5.16](#).

Table 5.16. Results from warm start tests performed on the oil-fired boiler.

Test Number	NO _x [g/GJ]	λ	Boiler Efficiency η (%)	Flue Temperature [C]
1	43.29	1.27	76.4%	79.9
2	45.23	1.36	73.7%	78.2
3	44.58	1.32	77.1%	79.6
4	50.38	1.37	79.4%	80.0
5	52.27	1.41	77.6%	79.6
6	57.18	1.54	79.4%	80.4
Average	48.82	1.38	77.3%	79.6
Uncertainty	10.7%	0.7%	8.9%	1.4%

As with the cold start tests, it was intended that the test conditions for each of the six tests would be as repeatable as possible, with the AFR as the main control variable. For the initial four tests, this was kept reasonably constant, whereas for the last two tests there was a shift towards leaner mixtures. The average NO_x emission factor was 48.82 g/GJ, which is slightly higher than for the previous operating modes, but the deviation between tests is quite small.

The next item to be investigated was the effect of the AFR on the NO_x emission factor. Seven separate tests were performed on the boiler, each with a different AFR setting. The boiler was allowed to settle into steady state operation for each test and then sampled for at least two hours. The results of these tests are presented in [Table 5.17](#).

Table 5.17. Effect of the relative air:fuel ratio λ on the NO_x emission factor

Test Number	NO _x [g/GJ]	λ
1	39.23	1.15
2	43.71	1.30
3	44.81	1.43
4	44.49	1.52
5	40.89	1.64
6	39.78	1.70
7	32.38	1.83
Average	40.75	1.51
Uncertainty	13.6%	0.8%

A plot of this data is shown in [Figure 5.9](#). The data was found to follow a quadratic trend, rising to a peak value of 44.8 g/GJ at 43% excess air before falling away as more air was added to the combustion chamber. This trend is very similar to Figure 2.6, in that the NO_x concentration reaches a peak when the mixture is lean and then decreases the AFR increases. The slight difference between the two graphs is that in the ideal case (presented in Figure 2.6), the NO_x peak occurs at approximately 5% excess air, whereas in this series of tests the peak was found to occur at 43% excess air.

There are a number of possible explanations for this difference, the most plausible being that the level of mixing of fuel and air in the burner prior to combustion was less than ideal. Of the total amount of air supplied to the burner through the air damper, only a certain fraction gets intimately mixed with the

fuel prior to combustion. It is this initial quantity of air that gets used in the reaction, and the remaining air does not mix with the fuel and, hence, is not used in the reaction. If the damper opening was reduced, so as to keep the total air input to the burner at a lower level, then the actual combustion mixture in the flame would become fuel rich and additional pollutants would be formed by incomplete combustion. However, the measured AFR would appear to be lean due to non-ideal fuel-air mixing prior to combustion. Indeed it was found in all tests that if the damper was closed, such that the measured AFR was less than approximately 1.25, then the flue gas began to show signs of a rich combustion mixture, i.e., increased levels of particulate matter and CO, indicating that even though the AFR was showing up to 25% excess air, combustion conditions within the flame were in fact fuel-rich.

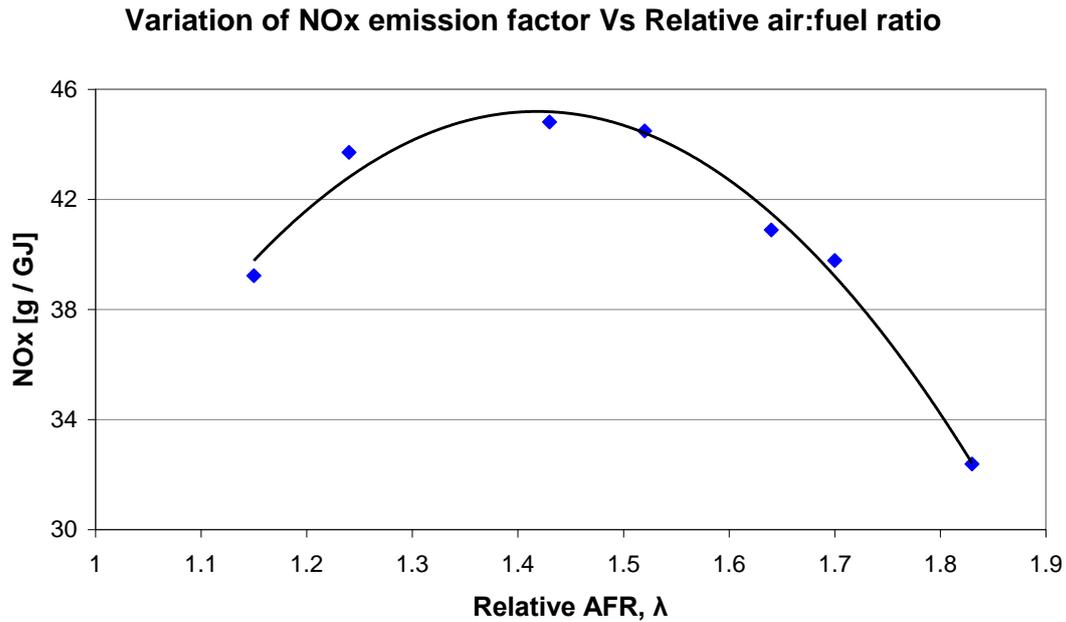


Figure 5.9. Plot of NO_x emission factor versus relative air:fuel ratio λ . The graph shows that the concentration of NO_x increases until it finds a maximum value at $\lambda=1.43$ (43%) excess air and then begins to fall away to lower values.

The final set of NO_x measurement tests performed on the oil boiler sought to investigate whether wear on the burner's atomisation nozzle dimensions had any effect on the level of pollutants emitted. As described in the section on PM emissions from the oil boiler ([Section 5.3.2](#)), two worn nozzles, which had been deemed to be past

their useful life by a service technician during a routine service were acquired and installed in the laboratory boiler. For each nozzle, the boiler was subjected to two steady state tests, keeping operating parameters, such as the AFR, as constant as possible, and the NO_x emission factor calculated. The results from these tests are presented in [Table 5.18](#).

Table 5.18. Results of tests to find the NO_x emission factor carried out using two old nozzles that were deemed to be worn out

Test Number	NO _x [g/GJ]	λ	Boiler Efficiency η (%)	Flue Temperature [C]
Old Nozzle 1				
1	36.65	1.35	95.3%	88.4
2	37.80	1.33	99.8%	88.5
Old Nozzle 2				
3	39.33	1.34	94.9%	79.0
4	41.06	1.33	96.9%	80.1
Average	38.71	1.34	96.7%	84.0
Uncertainty	12.6%	0.8%	7.5%	1.3%

In fact, the NO_x emission factor, as measured in these tests, was lower than the value found in the steady state tests on the brand new nozzle, and test conditions were almost identical for both test series. Hence, the level of wear appears to have no significant influence on the level of pollutants emitted to the atmosphere.

5.4.3 NO_x emissions from gas boiler

The results of the NO_x emission factors calculated during steady state operation of the gas boiler are given in [Table 5.19](#).

Table 5.19. Results from steady state tests performed on the gas-fired boiler

Test	NO _x [g / GJ]	λ	Efficiency [%]	Flue Temperature [C]	Sample Time [mins]
1	25.49	1.21	92.9%	60.9	197
2	25.35	1.22	90.7%	60.2	300
3	25.85	1.24	92.0%	61.3	394
4	26.57	1.23	92.5%	61.7	480
Average	25.81	1.23	92.0%	61.0	342.75
Uncertainty	18.9%	1.1%	8.0%	1.8%	

The measured NO_x emission factor values were almost identical from test to test and the final average was 25.81 g/GJ. As this is a condensing boiler, the flue gas temperature was very low when compared to older boilers that did allow the water vapour in the flue gas to condense. The dew point temperature for combustion of gas was calculated as 55°C, so while the amount of condensation that actually occurred during these steady state tests was very small, there was an increase in efficiency as less heat was being rejected to the atmosphere. The efficiency values listed here are based on the higher heating value, which takes into

account the latent heat of vapourisation of water that gets released during the condensation process.

The final pair of tests performed on the gas boiler examined its performance during both cold and warm start-ups. The cold start test period was defined as the first ten minutes of operation, and the test was repeated 31 times so as to collect a sufficient quantity of PM. The measured emission factor during cold start-up was 18.05 g/GJ for NO_x, while for warm starts the emission factor was found to be 16.75 g/GJ. The results of these two tests are shown in [Table 5.20](#).

Table 5.20. Results from cold and warm start transient tests performed on the gas boiler in the laboratory

Test	NO _x [g/GJ]	λ	Efficiency [%]	Flue Temperature [C]	Sample Time [mins]
Cold start	18.05	1.26	82.4%	45.0	310
Uncertainty	25.8%	1.1%	8.0%	2.5%	
Warm start	16.75	1.30	51.7%	50.4	297
Uncertainty	28.3%	1.1%	12.0%	2.2%	

5.4.4 Summary of NO_x measurements

All aspects of performance of a domestic oil-fired boiler were examined and relevant

pollutant emission factors measured at each operating condition. A brief summary of these emission factors is given in [Table 5.21](#) and [Table 5.22](#).

Table 5.21. Wood-pellet NO_x emission factors to be used for compiling emission inventories

Operating Mode	NO _x [g/GJ]
PL5	57.08
PL4	47.99
PL3	44.89
PL2	43.01
PL1	45.42
Startup	36.88

Table 5.22. NO_x emission factors measured on oil and gas boilers

	Oil	Gas
Test Condition	NO _x [g/GJ]	NO _x [g/GJ]
Steady State	42.82 ± 11.3%	25.81 ± 18.9%
Cold Start	44.77 ± 11.1%	18.05 ± 25.8%
Warm Start	48.82 ± 10.7%	16.75 ± 28.3%

5.5 Particle microscopy

A number of particle samples were analysed in the SEM to examine particle morphology and composition. Some interesting particle shapes were found, such as those seen in [Figure 5.10](#). In this case, the particles were observed to have a very regular crystalline shape and when the composition was found using EDX, the two elements present were

potassium and sulphur, indicating that the particles were crystals of potassium sulphate (K₂SO₄). The particles shown are very large in size, in the region of 10-20 µm, certainly a lot larger than the cut-point of the impactor stage. It is thought that the particles experienced hygroscopic growth when they initially landed on the substrate surface within the moist environment of the cascade impactor.

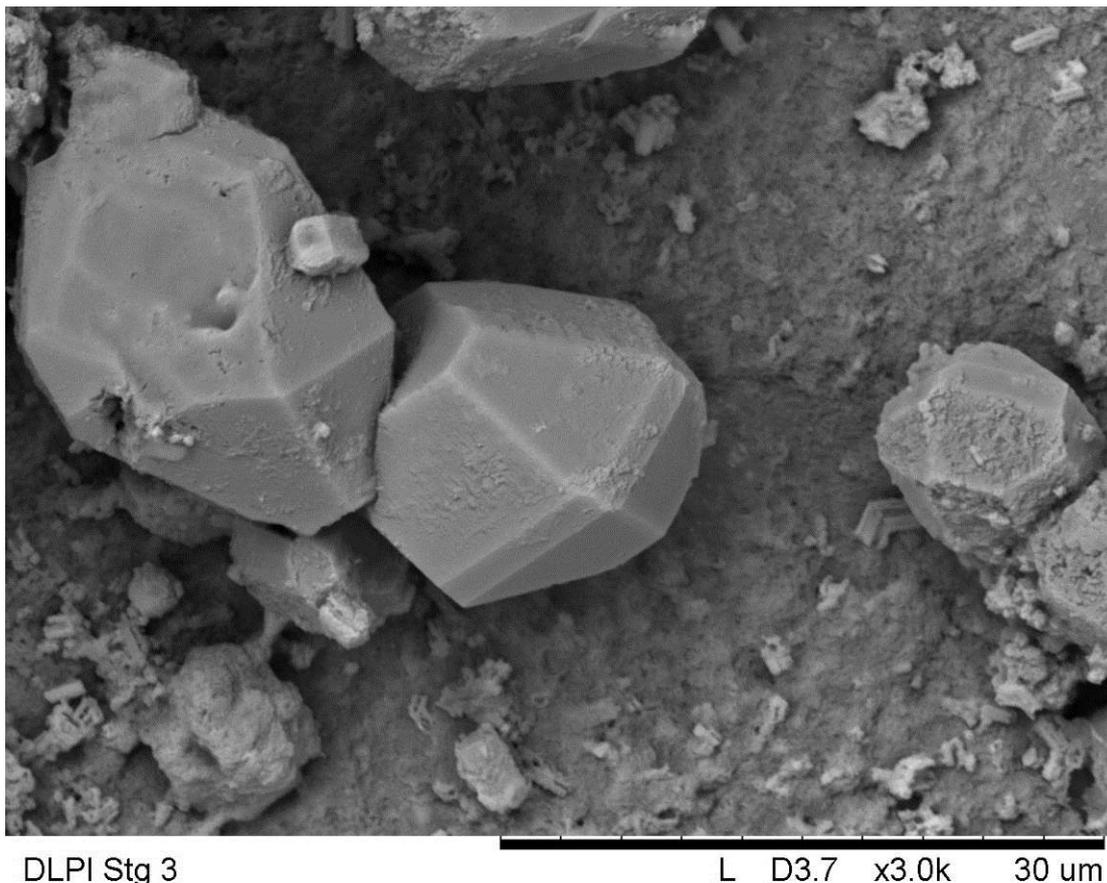


Figure 5.10. SEM image of a particle released from wood-pellet combustion and captured on Stage 3 of DLPI

A number of particle samples from the oil boiler were examined in the scanning electron microscope. The particles were found to be spherical in shape and ranged in size from just a few nanometres, right up to the micron scale. [Figure 5.11](#) shows a

micrograph taken of particles captured on Stage 6 of the DLPI. The particles were mainly composed of carbon, oxygen and silicon with trace amounts of sodium, magnesium, iron, sulphur and calcium.

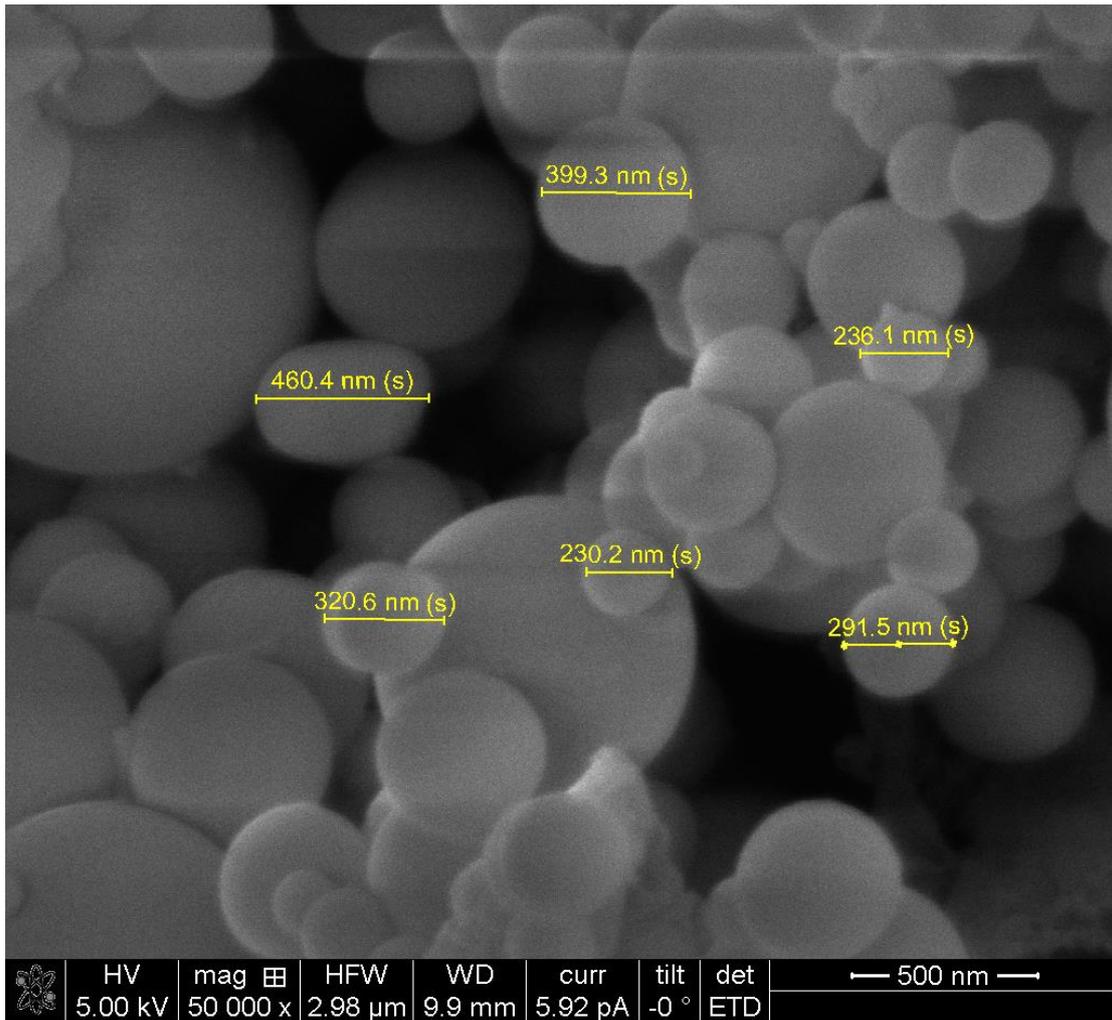


Figure 5.11. SEM micrograph taken from Stage 6 of DLPI. The particles consisted mainly of carbon, oxygen and silicon.

[Figure 5.12](#) shows a micrograph taken from a different stage in the DLPI. In this case, the particles are smaller in size than those seen in [Figure 5.11](#) because the cut point of Stage

3 is lower than for Stage 6. Once again, the particles were observed to be largely spherical in shape and consisted of the same elements as those seen in [Figure 5.11](#).

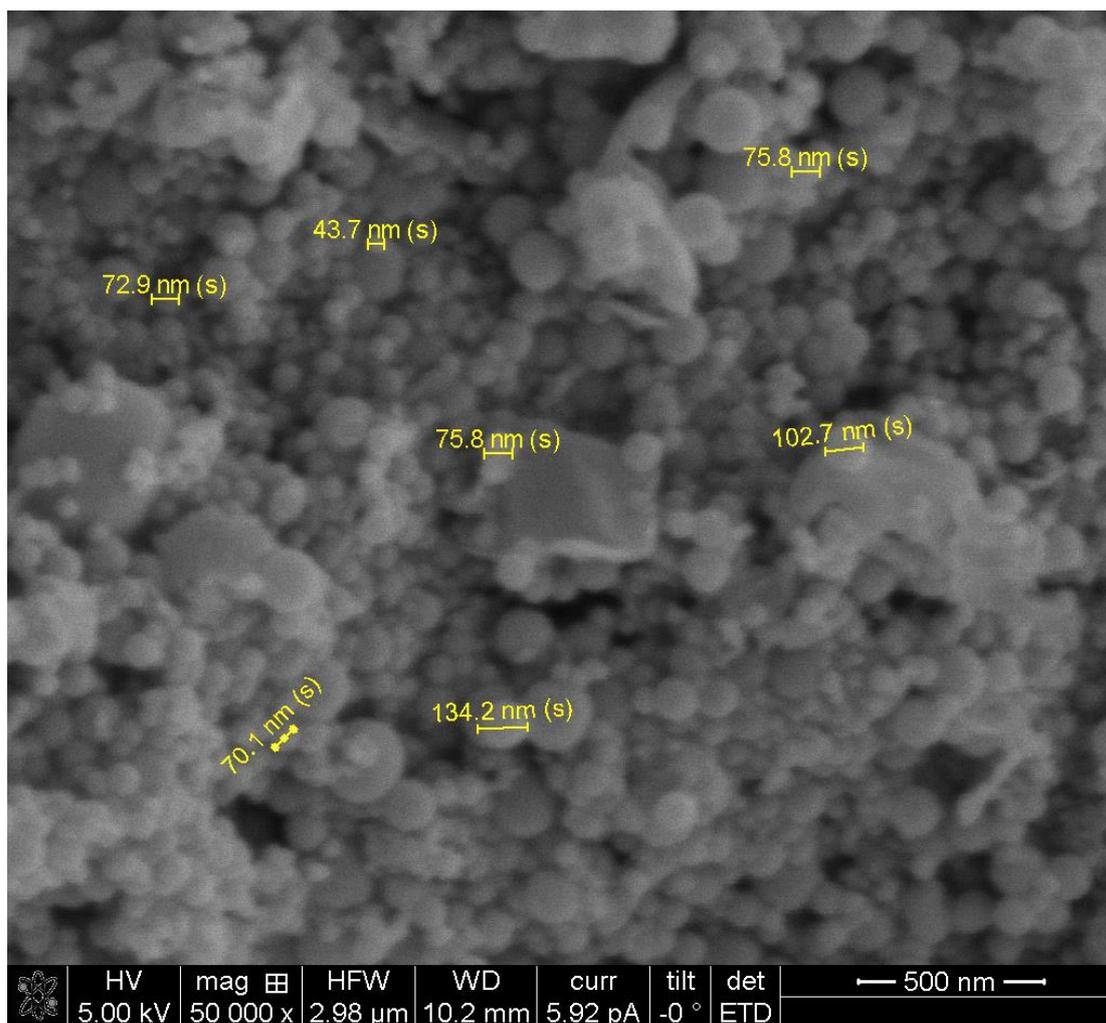


Figure 5.12. Particles captured on Stage 3 of the DLPI

5.6 Field testing campaign

Finally, as outlined earlier, a field measurement campaign was carried out to complement the tests performed in the laboratory. This involved visiting various installations and sampling the flue gas during steady state operation. As described previously, the experimental facility was built in the laboratory, as if the emissions testing had taken place entirely in the field, there would have been too many variables present to get any reliable results. Nevertheless, flue gas composition at a wide range of installations was measured in an attempt to see how emissions from a real-world boiler,

of varying ages and service levels, differed from those measured in the tightly controlled, modern, state-of-the-art boilers used in the laboratory. A total of 31 domestic boilers (23 oil-fired, six gas-fired and two wood-pellet) and 10 commercial/light industrial boilers (four oil-fired, five gas-fired, one wood-chip) were sampled. It was not possible to measure the particulate matter concentration from these boilers, as both the DLPI and filter sampling system were not portable. In addition, PM sampling required both close control on all operating parameters, so as to ensure constant repeatable test conditions, as well as knowledge of both the fuel and

flue gas flow rates, none of which were available in the field testing campaign.

The efficiency of these installations was measured using the 'loss method', whereby the rate of heat loss by various mechanisms is calculated and taken from 100%. In the results presented in this section, only the heat loss through the flue is taken into account, as there was no way to reliably measure heat loss by other mechanisms, such as losses through the boiler casing. These losses can be significant depending on the type of installation. For example, a boiler installed in an outhouse will generally operate in cold ambient temperatures, leading to high levels of heat loss to the surrounding air in the outhouse, whereas a boiler installed within a house will not be subjected to cold ambient air and have much lower levels of heat loss through the casing. Hence, the values quoted tend to overestimate the true boiler efficiency. As described in Chapter 3, a more accurate method of measuring the efficiency is to calculate the rate at which heat is added to the water in the circulation system by measuring flow and return temperatures and water flow rate and dividing it by the energy input to the boiler, based on the fuel flow rate. As none of these parameters could be measured in the field testing campaign, the loss method of measuring efficiency is the

only available option. The following sections outline the results obtained from sampling domestic and commercial scale installations.

5.6.1 Domestic field measurement campaign

The results of the field measurement campaign for domestic oil-fired boilers are given in [Table 5.23](#). A total of 23 samples were taken, at a variety of urban and rural locations and dwelling types. The variation between tests is represented as ± 1 standard deviation (68.3% confidence interval). For the NO_x emission factor, this variation is $\pm 21.9\%$, indicating a reasonably high degree of repeatability between tests. The average NO_x emission factor was found to be 35.632 g/GJ, some 16% lower than the steady state value of 42.815 g/GJ as measured in the laboratory. The average level of excess air was found to be 57% ($\lambda = 1.57$) but as it was not possible to control the air:fuel ratio during these tests, there was a high degree of variation between them (standard deviation = 37.1%). However, this figure is strongly influenced by three tests in particular (tests 13, 16 and 21), whose λ values are very high, suggesting that there may have been either an issue with the sampling equipment during that particular measurement or a leakage of air into the flue between the boiler exit and the sampling point, which distorted the gas composition being measured.

Table 5.23. Results from the field measurement campaign of oil-fired boilers

Sample No.	Location		Boiler Size [kW]	λ	Efficiency	NO _x [g/GJ]
1	Kildare	Rural	20	0.95	91.9%	27.94
2	Kildare	Rural	35	1.12	92.7%	26.26
3	Kildare	Rural	35	1.13	92.9%	26.03
4	Wicklow	Urban	32	1.30	90.0%	35.11
5	Wicklow	Urban	32	1.44	87.9%	29.37
6	Kildare	Rural	35	1.39	90.8%	34.23
7	Kildare	Rural	35	1.40	91.0%	34.97
8	Kildare	Rural	25	1.59	92.8%	25.19
9	Kildare	Urban	19	1.35	89.3%	39.49
10	Kildare	Rural	44	1.48	95.9%	26.60
11	Kildare	Urban	19	1.13	93.3%	39.02
12	Dublin	Urban	26	1.77	88.3%	31.85
13	Mayo	Urban	20	2.08	92.8%	44.69
14	Mayo	Rural	20	1.43	82.3%	33.01
15	Dublin	Urban	30	1.40	85.9%	39.31
16	Kildare	Rural	23	3.96	88.4%	35.35
17	Dublin	Urban	20	1.45	93.8%	46.47
18	Kildare	Rural	35	1.44	90.7%	36.73
19	Kildare	Rural	35	1.55	93.1%	31.86
20	Kildare	Rural	35	1.83	88.3%	55.86
21	Kildare	Rural	35	2.17	85.3%	34.02
22	Kildare	Rural	35	1.55	87.7%	34.88
23	Kildare	Rural	25	1.24	89.5%	51.30
		Average	29.1	1.57	90.2%	35.63
	Lab based values		28.8	1.30	96.7%	42.82
	Standard Deviation (%)		24.5%	37.1%	3.4%	21.9%

Table 5.24 presents the results obtained from tests performed on domestic gas-fired boilers. A total of six installations were tested, and all six dwellings were in an urban setting, reflecting the fact that the gas network is available in many urban centres, but only a limited number of rural areas have access. The average NO_x emission factor in this case was 48.30 g/GJ, almost twice as large as the emission factor measured in the laboratory. Again, the variability between the

six samples was reasonably small, with a standard deviation of 19.5%. As in the oil-fired samples, three boilers (Samples 3, 4 and 6) showed very high levels of excess air, suggesting that there could well be an inaccuracy in the reading, most probably due to air leakage into the flue. Indeed, for these three tests, it is highly unlikely that the λ readings are true because running over 100% excess air would extinguish the flame.

Table 5.24. Results from field measurements performed on gas-fired boilers

Sample No.	Location		Boiler Size [kW]	λ	Efficiency	NO _x [g/GJ]
1	Dublin	Urban	15	1.23	91.9%	34.97
2	Dublin	Urban	20	1.69	89.1%	50.98
3	Dublin	Urban	25	2.42	86.7%	63.74
4	Dublin	Urban	26	3.00	85.6%	40.24
5	Dublin	Urban	11	1.55	92.6%	54.47
6	Dublin	Urban	15	2.16	87.0%	45.42
		Average	18.7	2.0	88.8%	48.30
		Standard Deviation (%)	29.5%	29.4%	3.0%	19.5%

The final samples taken in the domestic sector were from wood-pellet boilers. As these are a new technology in Ireland, and are not yet widespread across the country, finding suitable installations to sample from was more difficult and, hence, only two wood-pellet boilers were sampled in the field. The first boiler tested was designed both as a stove and central heating boiler. It was installed in the living space of a house and, in addition to heating water in its boiler, which could subsequently be used in the

central heating system, it also acted as a room heater. As the efficiency calculation used does not take heat loss through the boiler casing into account, the efficiency value quoted here is only measuring one aspect of the boiler's performance. This particular boiler was not as technologically advanced as the Gerkros boiler that was used in the laboratory testing series and, as a result, the NO_x emission factor of 74.84 g/GJ is considerably higher than the value of

57.08 g/GJ found during steady state operation in the laboratory ([Table 5.25](#)).

The second boiler sampled was the same model as the one used in the laboratory and, in this case, the NO_x emission factor was in

close agreement with the value measured in the laboratory. As only two boilers were sampled, and one was more technologically advanced than the other, the variation between the two test results is not included, as the sample size is too small.

Table 5.25. Results from field measurements performed on wood-pellet boilers

Sample No.	Location		Boiler Size [kW]	λ	Efficiency	NO _x [g/GJ]
1	Dublin	Rural	25	3.21	77.6%	74.84
2	Wicklow	Rural	25	1.83	96.1%	45.79
		Average	25	2.5	86.9%	60.31

5.6.2 Commercial field measurement campaign

In addition to the domestic samples, a series of tests were performed on larger scale, commercial/light industrial boilers. These boilers would typically be used to heat office buildings, schools, shopping centres, etc., and range in size from approximately 50kW to 500kW. In total, four oil boilers, five gas boilers and one wood-chip boiler were tested.

The results for the four oil tests are given in [Table 5.26](#). The measured NO_x emission factor was 29.94 g/GJ, which is less than the emission factors measured in the laboratory (42.82 g/GJ) and in the domestic field campaign (35.63 g/GJ). It is worth noting that the two boilers sampled in Monaghan (Samples 1 and 2) were faulty on the day the tests were performed. The excess air level was very high and yet there was over 2000ppm of carbon monoxide (CO) in the flue gas, indicating a high degree of

incomplete combustion. It is thought that the atomising nozzles in the two burners were damaged and that the fuel was not being properly atomised, and that this resulted in incomplete combustion and the consequent high levels of CO. In addition, there was poor mixing of air with the fuel and this resulted in the AFR appearing very large. As a result of these issues, as well as very high flue gas temperatures, the efficiencies of the two Monaghan boilers were both very low. Hence, there is significant doubt over the validity of the NO_x emission factors measured from those two boilers as they were not a representative example of a boiler operating correctly.

The two samples taken in Mayo (Samples 3 and 4) are a much more reliable estimate of the true NO_x emission factor from a typical large-scale oil-fired boiler. Sample 3 was taken from a 12-year-old boiler, whereas Sample 4 was from a boiler less than a year old. The newer boiler had a considerably

higher efficiency, and the rate of heat loss through the flue was much less, as evidenced by the lower flue gas temperature. Both boilers were well serviced and the AFR was set correctly. The average NO_x emission

factor for these two boilers was 34.67 g/GJ, which is very similar to the 35.63 g/GJ measured during the domestic field campaign.

Table 5.26. Summary of results from samples taken from commercial scale oil-fired boilers

Sample No.	Location	Building Type	Boiler Size [kW]	λ	Efficiency	Flue Temperature [C]	NO _x [g/GJ]
1	Monaghan	Office Building	120	2.27	79.4%	260.1	20.51
2	Monaghan	Office Building	120	2.27	79.2%	264.0	29.90
3	Mayo	Office Building	332	1.25	88.5%	249.1	36.60
4	Mayo	School	250	1.31	92.4%	162.7	32.75
Average			205	1.77	84.9%	234.0	29.94
Standard Deviation (%)				27.9 %	6.8%	17.7%	19.8%

The five gas boilers tested were all used in EPA regional office buildings and the results are given in [Table 5.27](#). The NO_x emission factor for these boilers was measured as 19.01 g/GJ, approximately 25% less than the 25.81 g/GJ value measured in the laboratory. One important parameter to note is the flue gas temperature, which in all cases is considerably lower than the temperatures measured in the oil-fired boilers. Samples 1 and 2 were both

condensing boilers, and as the dew point for natural gas combustion products is approximately 55°C, it is clear that these two boilers were operating at or close to condensing mode, and this is reflected in their very high efficiency values (based on lower heating value). The other three boilers were non-condensing but still operated at high efficiency, mainly because the heat loss through the flue was very low.

Table 5.27. Summary of results from tests performed on commercial scale gas-fired boilers

Sample No.	Location	Building Type	Boiler Size [kW]	λ	Efficiency	Flue Temperature [C]	NO _x [g/GJ]
1	Dublin	Office Building	65	0.89	97.6%	56.1	13.06
2	Dublin	Office Building	145	1.25	96.0%	72.4	10.84
3	Wexford	Office Building	500	0.99	93.6%	126.1	20.32
4	Kilkenny	Office Building	250	1.31	93.6%	117.2	24.69
5	Kilkenny	Office Building	310	1.31	94.1%	110.1	26.16
Average			254.0	1.15	95.0%	96.4	19.01
Standard Deviation (%)				15.4%	1.7%	28.2%	32.2%

The final boiler to be tested in the field measuring campaign was a 400 kW wood-chip boiler used in the EPA's regional office in Wexford. The results of this test are shown in [Table 5.28](#). Wood-chips are a lower grade fuel than wood-pellets with a higher moisture content and lower energy density. Wood-chips typically have a moisture content of 30-40% and an energy density of 3.1 GJ/m³, while pellets have a moisture content of less than 10% and corresponding energy density of 11 GJ/m³. However, wood-chips are a much cheaper fuel and require significantly less energy to manufacture than wood-pellets. They tend to have higher ash contents and are preferred in large scale combustion appliances, such as those found in the commercial and industrial sectors, while pellets are preferred for domestic appliances where less space is available for fuel storage. Pellet manufacturing is an energy-intensive

process, as the raw material needs to first be dried then processed in a hammer mill to produce fine sawdust particles. Caserini *et al.* [127] have performed a life cycle analysis of the pellet manufacturing process, from tree felling to wood processing to fuel transport to the end use point. They found that the most energy-intensive process was the drying and conditioning of the raw material, which accounted for over 60% of total energy consumption over the life cycle analysis. Overall, it was found that the entire pellet manufacturing process required an energy input of 3 MJ/kg, or 17.8% of the pellet calorific value of 16.7 MJ/kg, meaning that in spite of the fact that significant inputs of fossil fuel derived energy are required in the production of biomass pellets, there is still a large net benefit in terms of greenhouse gas reductions.

Hence, a direct comparison of emission factors from combustion of wood-chip to those from combustion of wood-pellets is not possible. The average NO_x emission factor measured during the domestic campaign was 60.31 g/GJ, 26% lower than the wood-chip emission factor of 81.86 g/GJ. The

wood-pellet NO_x emission factor measured in the lab during steady state conditions was 57.08 g/GJ, close to the domestic value, but considerably lower than the wood chip emission factor. The large difference can be attributed mainly to the lower quality fuel resulting in higher emission levels.

Table 5.28. Results from sample taken from a 400kW wood-chip boiler used to heat an office building

Sample No.	Location	Building Type	Boiler Size [kW]	λ	Efficiency	Flue Temperature [C]	NO _x [g/GJ]
1	Wexford	Office Building	400	2.03	92.3%	115.7	81.86

5.7 Weighted average emission factor

Finally, the overall aim of the testing process was to derive an emission factor that can be used to generate more accurate emission inventories. When compiling inventories, it is not practical to consider a wider population of installations and deal with the minutiae of each duty cycle. Instead, it is necessary to have a single weighted average emission factor that allows for the various different duty cycles. By combining the results of the duty cycle analysis performed in this study with the laboratory measured emission factors, it is possible to arrive at a weighted average for each of the boilers.

For each installation, the total duration for cold and warm start-ups, steady state and modulating operation was found from the duty cycle analysis at each of the thirteen

locations. The quantity of fuel (in GJ) used by each boiler was calculated based on the burner's nominal thermal input rating. Using the emission factors measured in the lab, the total quantity of emissions produced at each operating state could then be calculated. This process was repeated for all deployments to find the total mass of PM and NO_x and total quantity of fuel used during the whole sampling period. The emission factor was then found by dividing the total mass of pollutant by the total quantity of fuel used. The weighted average emission factors for each of the three boiler types in question are given in [Table 5.29](#). The emission factors recommended for use by EMEP are also included as a comparison. It is clear from the results that the emission factors measured during this study are considerably lower than the EMEP values that are currently in use.

Table 5.29. Summary of weighted average emission factors to be used when compiling emission inventory reports. Emission factors from the EMEP Guidebook [128] are included as a comparison.

	Measured		EMEP Guidebook Value	
	PM [g/GJ]	NO _x [g/GJ]	PM [g/GJ]	NO _x [g/GJ]
Wood-pellet	22.4	49.8	76	90
Oil	0.31	44.13	3	70
Gas	0.03	32.73	0.5	70

6. Results: Non-road Transport

The current emission inventory, as compiled by the EPA for non-road transport, uses a Tier 1 approach. The aim of this study was to improve this to as high a tier as possible. For domestic aviation, a Tier 3 inventory was developed, while for international flights a Tier 3 analysis was not possible, as more information was required about total journey length, so a Tier 2 approach is used instead. Aircraft engine manufacturers measure particulate concentration using the smoke number method. Using this method, a sample of the particle laden exhaust gas is drawn through a filter paper and the consequent change in paper colour is compared to an unladen filter. The higher the PM concentration, the darker the filter paper becomes. However, no reliable method has been developed to correlate smoke number with an actual number or mass concentration, so at this stage it is not possible to accurately develop a PM emission inventory for aircraft. Therefore, the following results section will focus on NO_x only.

The first step was to analyse air traffic movements in Ireland, and detailed information was obtained from the Irish Aviation Authority. This information was from 2007 and included details of each flight that took place, whether they were arrivals or departures, and the engine type used in the aircraft. Information on over 289,000 flights was analysed, and details of traffic through Dublin, Cork, Shannon, Kerry and Knock airports was included. A Tier 2 method takes details of the LTO cycle into account when calculating total emissions. Information was

gathered relating to the total number of flights made by a specific aircraft. A total of 223 aircraft operated in Ireland in 2007 but of these just 30 accounted for over 90% of total movements, so in order to simplify the analysis, these 30 aircraft were put into a generic grouping. The calculated emissions based on these 30 craft were then extrapolated to include the remaining 10% of other flight movements. The next step was to combine the number of domestic flights made by each of the 30 aircraft, with information about LTO fuel consumption and NO_x emissions, to find the total LTO fuel consumption and NO_x emissions. The amount of fuel used during cruise is found by subtracting the LTO fuel from the total fuel used in the sector, as determined by the SEAI energy balance [15]. Once this cruise fuel consumption is found, the cruise emissions are found by simply multiplying it by the cruise emission factors, as found on manufacturer data sheets. This process is then repeated to find the NO_x emissions from international flights.

[Figure 6.1](#) shows the different LTO emission factors relating to both international and domestic flights. Emission factors published by the International Civil Aviation Authority (ICAO) are included as a comparison. The values shown for the three tiers are published in the 2009 version of the EMEP Guidebook, while the EPA emission factors are the ones used in the last inventory report and were taken from the 2002 Guidebook. The first point to note is that international LTO emissions are greater than their domestic counterparts, which is due to the

fact that international aircraft are significantly heavier than domestic, which means that they need greater thrust to get off the ground. It is clear for all sectors that the Tier 1 emission factor is considerably higher than the Tier 2/3 value. As Tiers 2 and 3 are used for more accurate inventories, it can be concluded that a Tier 1 approach

overestimates NO_x levels. The graph also shows that the current emission factors used by the EPA are overestimating domestic emissions by up to 30%, with even larger overestimates being seen in the international aviation sector.

Comparison of Domestic and International Emission Factors

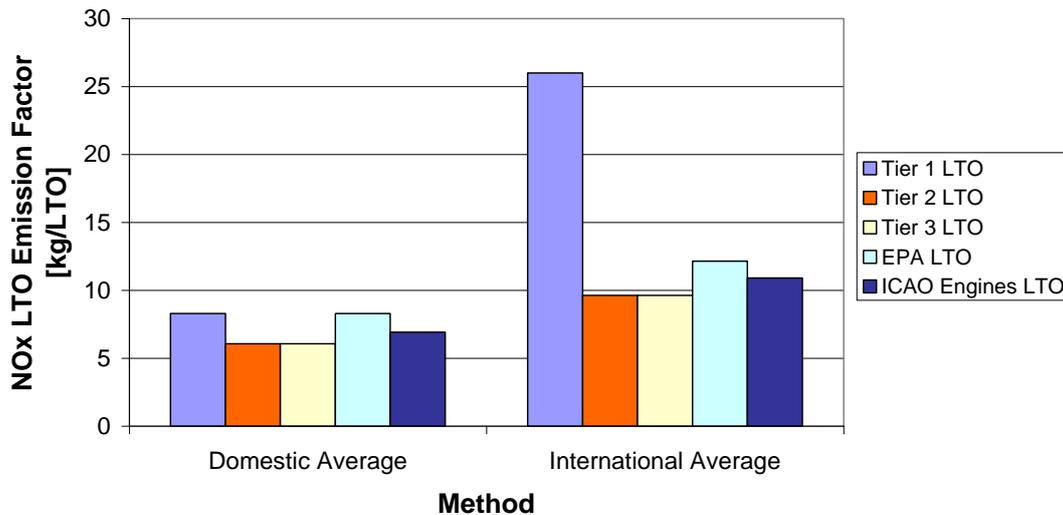


Figure 6.1. International Vs Domestic emission factors

For cruise activities, the emission factor used has a slightly different basis. Whereas LTO emission factors are given as a mass of pollutant per LTO, during cruise activities the emission factors are based purely on fuel consumption. So the accuracy of cruise emission inventories depends greatly on the accuracy of the calculated fuel consumption. The Tier 1 methodology uses standard LTO consumption figures for an aircraft and sums these for the total amount of flights. This is in turn subtracted from the total amount of fuel, from the SEAI energy balance, and the difference taken to be the cruise fuel consumption. However, when a Tier 3 analysis was used, it was found that the level of fuel consumption was being over-

estimated by the EPA. According to the energy balance, approximately 38,000 tonnes of jet kerosene was consumed in 2007, yet the Tier 3 results predict that just 23,000 tonnes were necessary. Granted, the Tier 3 analysis calculates a theoretical amount but a deviation of 70% seems highly unlikely. When the analysis is repeated using ICAO emission factors, the fuel consumption figures work out almost identical to the Tier 3 analysis so it can be taken that current fuel consumption estimates, and, as a result, emission levels are being overestimated. Estimates of total emissions from domestic and international aviation are shown in [Table 6.1](#)

Table 6.1. Aviation emission estimates

Tier 1		
	Total Domestic NO _x (Gg)	Total Domestic PM2.5 (Gg)
Domestic fleet	0.395	0.006
International fleet	12.906	0.168
Tier 2		
	Total Domestic NO _x (Gg)	Total International NO _x (Gg)
Average all fleet	0.428	12.12
Tier 3		
	NO _x (Gg)	
Total Domestic	0.341	
EPA		
	NO _x (Gg)	PM2.5 (Gg)
Domestic	0.260	0.008
International	6.643	0.176

The next results section relates to emissions from the rail sector. [Figure 6.2](#) shows the comparison of all of the emission factors for the different methods that were used in each of the estimates. It was not possible to

perform a Tier 3 analysis for rail transport, as this requires information regarding the total time of use for each train type, which was not available for this study.

Rail Emission Factors

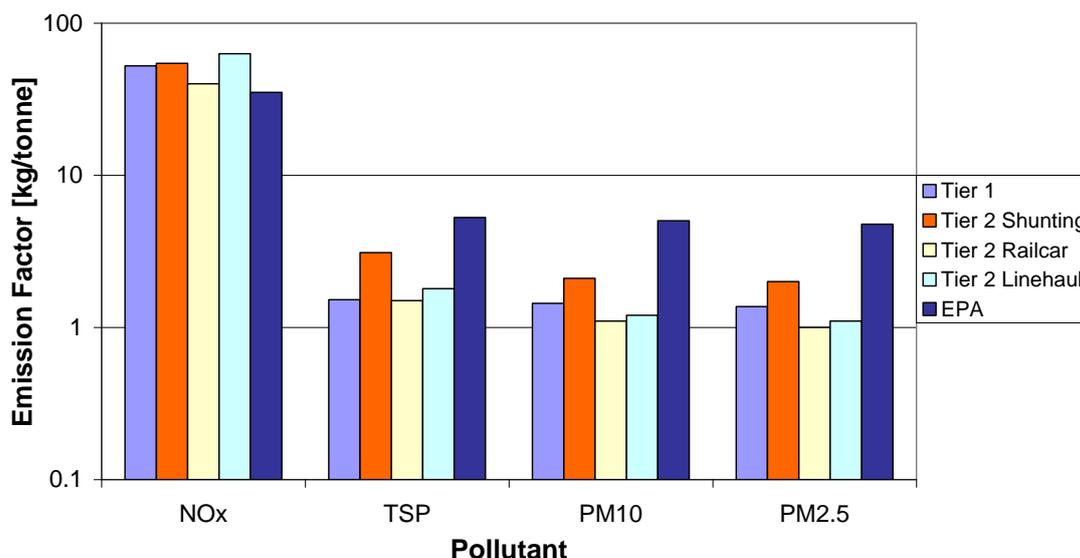


Figure 6.2. Rail emission factors

The first point to note is the relationship between the NO_x emission factors and then also the varying size of the particulate matter emission factors. As can be seen in the EPA column and in [Table 6.2](#), the NO_x emission factors are less than the tier emission factors, and also the EPA particulate emission factors are considerably larger than any of the other guidebook tier figures. It was found that the EPA NO_x estimate was up to 1.5 times smaller than the value found by either the Tier 1 or Tier 2 analyses, so that

the current emission inventory underestimates true NO_x levels from the rail sector. In contrast, however, the estimates of particulate matter emissions were considerably lower when both the Tier 1 and Tier 2 analysis was used. Each of the three emission factors for TSP, PM₁₀ and PM_{2.5} range from twice to three times that of the guidebook emission factors. This, in turn, affects the final results, with a major overestimation of rail PM emissions in the current emission inventory.

Table 6.2 Rail emission estimate results

Tier 1				
Train Type	Total NO _x [Gg]	Total TSP [Gg]	Total PM ₁₀ [Gg]	Total PM _{2.5} [Gg]
All	2.025	0.059	0.056	0.056
Tier 2				
Train Type	Total NO _x [Gg]	Total TSP [Gg]	Total PM ₁₀ [Gg]	Total PM _{2.5} [Gg]
All	1.792	0.061	0.044	0.04
EPA Inventory				
Train Type	Total NO _x [Gg]	Total TSP [Gg]	Total PM ₁₀ [Gg]	Total PM _{2.5} [Gg]
All	1.355	0.204	0.194	0.184

The final part of the non-road transport sector is that of sea transport, or navigation. This sector was the most troublesome area to perform a higher tier level estimate due to a lack of available information. After consultation with the relevant port authorities, detailed information on all shipping movements in Ireland in 2007 was obtained. Information on each of the vessels and their associated emission factors was taken from the Lloyds maritime information

service database. Two types of fuel are used for sea transportation: residual fuel oil and gas/diesel oil. Residual fuel oil is a much thicker, lower quality fuel than gas oil and tends to be used in larger engines, such as freight ships or car ferries. A comparison of emission factors for NO_x and PM from each of the fuels as used in the Tier 1 analysis and by the EPA in their inventory report is shown in [Figure 6.3](#).

Comparison of Navigation Emission Factors

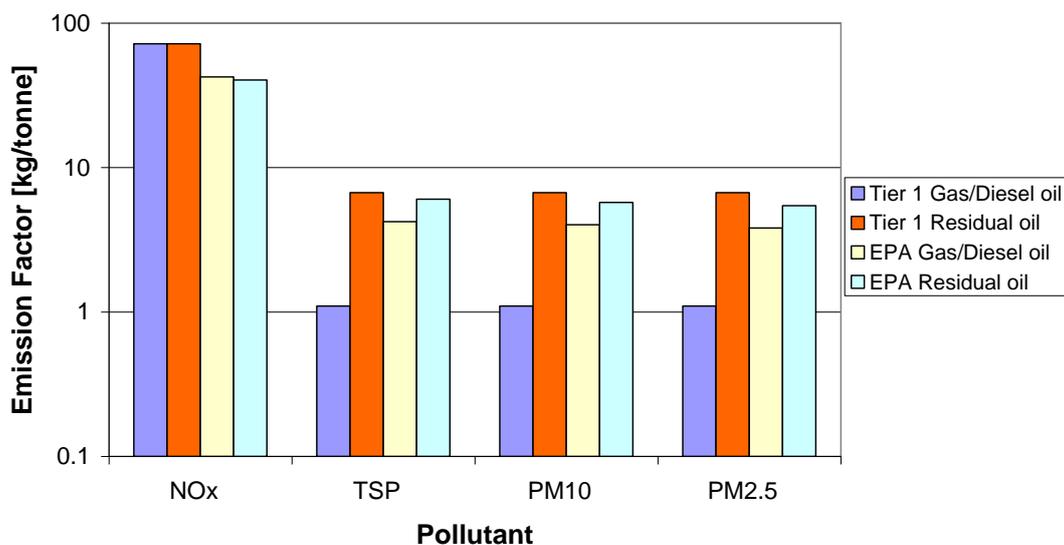


Figure 6.3. Comparison of navigation emission factors

Similar to the rail NO_x emission factors, the value used by the EPA in the inventory report is a lot lower than the emission factor used in the Tier 1 analysis, 40 kg/tonne versus 72 kg/tonne respectively. For the Tier 1 analysis, the NO_x emission factors were identical for both types of fuel and the PM emission factors are assumed to be the same for all varieties of particulate matter (PM₁₀, PM_{2.5} and TSP). It is also clear from the chart that residual fuel oil emits a much larger amount of PM than gas/diesel oil, which is to be expected, as residual oil contains greater levels of impurities following distillation due to being a bottom-end fuel type. However, the emission factors used for

PM by the EPA are considerably higher than the Tier 1 values. [Table 6.3](#) shows the estimates of total NO_x and PM from Irish navigation in 2007 using the Tier 1 and Tier 2 methods, as well as the values reported by the EPA. The total quantity of NO_x released from marine transport was 8.249 Gg, while the value calculated by the EPA and quoted in the inventory report was just 4.726 Gg, over 40% lower. Meanwhile, for PM, the value calculated by the EPA was 0.606 Gg for TSP while the Tier 1 method estimates that TSP levels were 0.502 Gg, indicating that the EPA was overestimating PM by 20%.

Table 6.3. Navigation emission estimate results

Tier 1				
Train Type	Total NO _x [Gg]	Total TSP [Gg]	Total PM ₁₀ [Gg]	Total PM _{2.5} [Gg]
All	8.249	0.502	0.502	0.502
EPA Inventory				
Train Type	Total NO _x [Gg]	Total TSP [Gg]	Total PM ₁₀ [Gg]	Total PM _{2.5} [Gg]
All	4.726	0.606	0.575	0.547

The next stage was to examine marine emissions using the Tier 2 method. A number of major assumptions had to be made due to large gaps in the available data. As a result, the calculated emissions estimate could not be split up into domestic and international navigation. For the Tier 2 method, there are four different ship categories to include:

- Gas turbine
- High-speed diesel
- Medium-speed diesel
- Slow-speed diesel.

The associated emission factors are given in [Figure 6.4](#) for each engine type and then respective fuel type.

Comparison of ship engine NOx Emission Factors

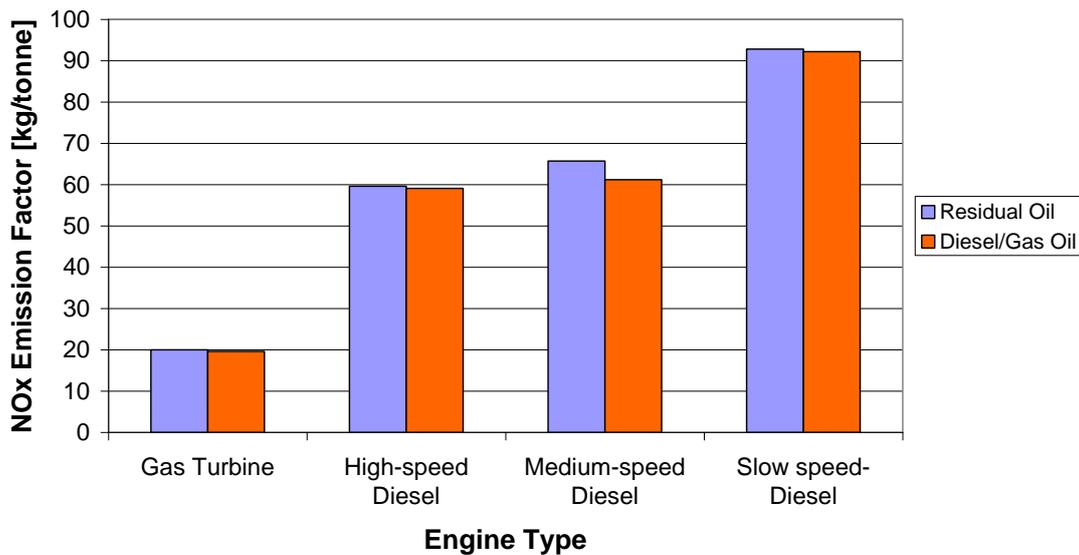


Figure 6.4. Emission factors for generic engine types

It is interesting to see the huge variation between the emissions from the gas turbine engine compared to the slow-speed diesel engine. A difference of over four times the NO_x emissions per tonne of fuel being consumed is found to occur. An important note can be made about the comparison between the different tier methods in contrast to aviation. During the higher tier aviation methods, the fuel consumption figures become independent of the energy balance and therefore two dependent features exist, the emission factors and the fuel consumption of each specific aircraft. All estimates are based purely on the given fuel consumption in the energy balance. No independent fuel consumptions are derived. This means that the comparisons between

the different tier methods with the sector of navigation transport are completely reliant on the emission factors put in place.

On looking at the total tonnage of fuel consumed by shipping within Ireland, it is found that over 60% is associated with slow-speed diesel engines. The reason for this is that these engines have a high efficiency (also at part-load) and they can run on heavy fuel oil, which is cheaper than other fuel types such as gas oil. This is a very important observation because, as mentioned previously, the slow-speed engine has the highest emission factors associated with it, which leads to greater emissions than if the fuel was used in a medium or high-speed diesel engine.

7. Results Summary

As stated earlier, the aim of this study is to improve Ireland's emission inventories for NO_x and PM from small combustion installations (SCIs) in Ireland. The current inventory report is compiled by the EPA Office of Climate, Licensing and Resource Use (OCLR). [Figure 7.1](#) and [Figure 7.2](#) show the quantities of NO_x and PM as reported in the emission inventory report and compares

them to the re-calculated total using the emission factors developed in this research. The contribution of the road transport sector to overall emissions is included as a comparison. As this work package does not examine the road transport sector, the total quantity of road transport emissions for 'This Study' remains the same as the reported amount.

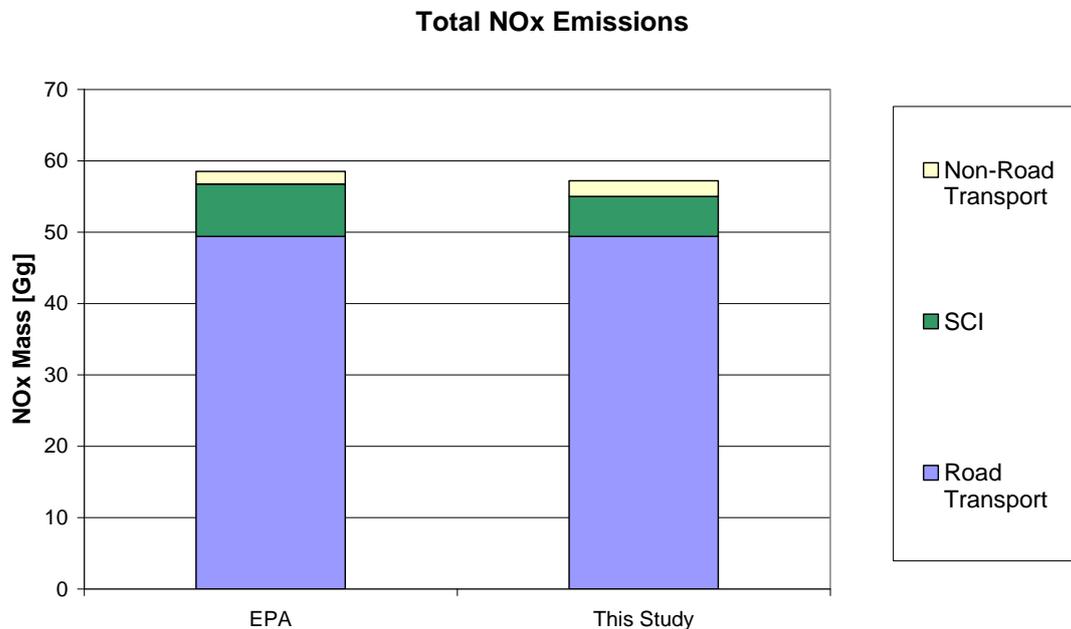


Figure 7.1. Comparison of total NO_x emissions for the road transport, non-road transport and small combustion installation sectors as reported in current inventory and the recalculated using emission factors measured in present study

The first point to note in [Figure 7.1](#) is that road transport completely dominates the mass of NO_x emissions in Ireland. A reduction of almost 25% was found for small combustion installations, driven primarily by the difference in emission factors, as outlined earlier in this report. NO_x emissions

from the non-road transport sector are currently being underestimated by approximately 20% but, as [Figure 7.1](#) shows, this sector makes only a tiny contribution to overall NO_x levels when compared to road transport and SCIs.

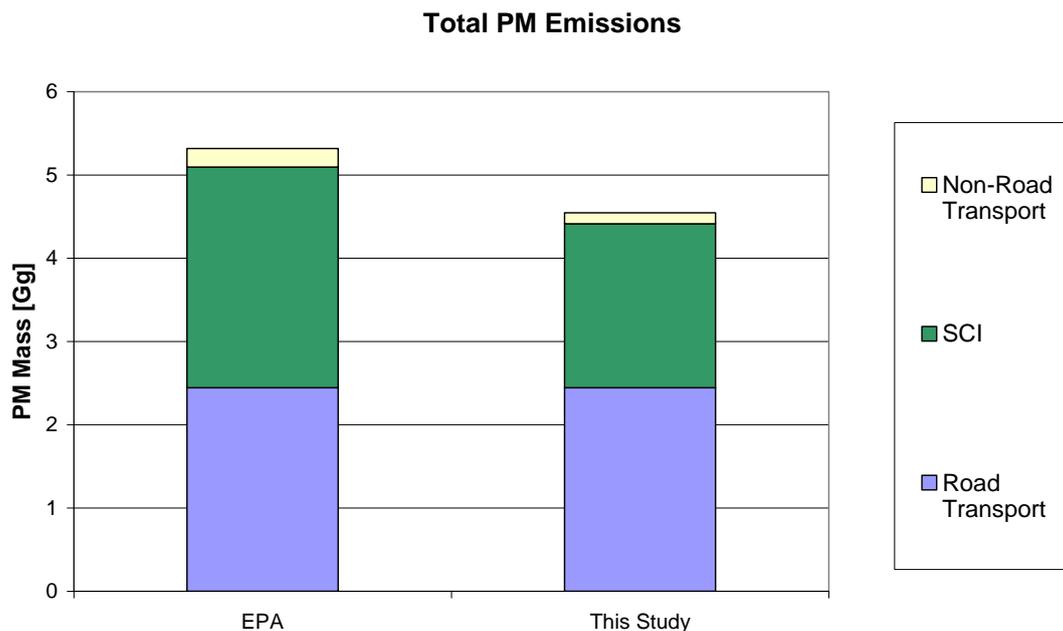


Figure 7.2. Comparison of total PM emissions for the road transport, non-road transport and small combustion installation sectors as reported in current inventory and the recalculated using emission factors measured in present study

For PM, however, the road transport sector does not make as significant a contribution to overall levels as it does for NO_x . In fact, the SCI sector contributes approximately the same amount of PM as the road transport sector, and this study has found that the current EPA emission inventory overestimates PM from this sector by almost

35%. Similarly, for non-road transport, the calculated emission levels were found to be 40% lower than the reported amount, although this change makes very little impact on the overall emission levels, as this sector is very small when compared to either SCIs or road transport.

8. Discussion

PM Test Results

The results from the particulate matter testing series were very interesting. Taking the case of the wood-pellet boiler to begin with, it was found that the operating condition or power level of the boiler had a significant effect on the PM emission factor. During operation at the nominal heat output, the emission factor was measured as 14.6 g/GJ, while during a cold start-up, a peak of 68.2 g/GJ was reached, an almost five-fold increase. Even during this cold start phase when combustion conditions were at their worst, the measured emission factor was still less than the value of 76 g/GJ recommended for use in the EMEP Guidebook.

In fact, the discrepancy between the values in the EMEP guidebook and those measured in the laboratory was a recurring theme during this project. Why the EMEP values are so much higher is not clear. When examined more closely to find EMEP's emission factor sources, it transpired that numerous sources were reports written in Poland, in Polish and were not available for scrutiny. One possible explanation is that in Poland and other parts of Eastern Europe, the climate is much colder in winter time and, consequently, their heating season lasts for far longer than the Irish equivalent. Hence, technologies employed in heating appliances tend to be very different as well. In these colder climates, heating systems are designed to run for very long periods of time providing a constant baseload supply to a building, which is probably very well

insulated. In Ireland, however, our buildings traditionally have poor levels of insulation and with our milder winters our demand for heat is less. Hence, boilers tend to run intermittently and burn more intensely while they are running resulting in lower emissions concentrations.

During the wood-pellet boiler tests, the PM emission factor increased as the boiler output dropped. It was found that as the operating load on the boiler decreased, there was a corresponding increase in the level of excess air, represented by the relative air:fuel ratio λ . In addition, the concentrations of particulate matter and carbon monoxide (CO) also increased as the level of excess air increased. This phenomenon has been observed before by Johansson *et al.* [108] and can be explained as follows. CO is an unoxidised compound and is a consequence of imperfect combustion conditions. However, a higher λ value corresponds to higher concentrations of CO, so it is clear that oxygen is not the limiting parameter for CO oxidation during poor combustion conditions. In reality, the excess air cools the combustion chamber leading to incomplete combustion. Hueglin *et al.* [129] also noted that a higher level of excess air resulted in a higher concentration of particulate emissions, and that the level of air supply had a larger influence than any other parameter on the particulate matter concentration from combustion of wood fuels.

It is also important to note that the PM concentration increases dramatically during the start-up phase. As has been suggested by Wiinikka [130], the cold combustion chamber found during start-up can lead to flame quenching at the burner wall, and increased level of particulate matter being formed. As the flame developed from the initial ignition point, and as the temperature in the burner increased, the particulate concentration began to decrease before settling at a value similar to the concentration found when the boiler was operating at nominal capacity. This highlights the contribution of the duty cycle of the boiler to the overall concentration of PM emitted to the atmosphere, and is one of the main reasons why pellet boilers are designed with modulating burners, so as to avoid frequent stop-start operation.

Particle mass size distribution from combustion of wood-pellets was also measured and was found to be dominated by a sub-micron mode centred about an aerodynamic diameter of 190 nm. Almost 90% (by mass) of the PM had a diameter of less than 1µm but a small super-micron mode was also found around 6 – 8 µm in the startup and power level 1 tests. A selection of sampled particulates was examined using a scanning electron microscope (SEM) and energy dispersive x-ray analysis (EDX). Three distinct types of particle were found:

- Large grain like particles. These were found to contain mainly potassium and sulphur and, hence, are assumed to be K₂SO₄ crystals
- Agglomerate of small particles clustered around larger crystalline particles. These were found to contain potassium, chlorine and

sulphur and hence consist of K₂SO₄ and KCl crystals.

- Large spherical particles containing calcium and potassium, indicating that they were formed by fly ash particles being ejected from the combustion bed.

The formation mechanisms for these alkali compounds (K₂SO₄ and KCl) has been dealt with in detail by Jiménez *et al.* [131], who found that when flue gas temperature dropped below 1000°C, KOH and SO₂ reacted to form K₂SO₄ crystals, and that below 800°C KCl, condensed on the surface of the K₂SO₄ crystals. As temperature dropped further, they found that particle size continued to grow due to continued condensation from the gas phase.

Three operating modes were examined for the oil boiler: cold start, steady state and warm start. PM was sampled using both the DLPI and a filter sampling system. The emission factor measured using the filter system was taken as the definitive value for the oil boiler, as a number of technical issues with the DLPI, in addition to the very small mass of particles being collected on each stage, meant that there was a more significant uncertainty about the accuracy of the DLPI-measured mass concentration. Instead, the DLPI results are used solely as a source of size distribution data.

The results of these tests were somewhat surprising. Initially, it had been thought that start-ups and transients would produce a higher concentration of particulate matter than steady state operation, due to lower flame temperatures and sub-optimal combustion conditions. However, as the results have shown this was not the case.

The PM concentration from the transient tests was actually lower than that from steady state operation. One possible explanation for this is that when the boiler was operating in condensing mode during start-ups, it was reducing PM concentration by condensing onto existing particles and washing them out of the exhaust stream, as described in the literature review (Section 2.4) [31, 33, 34].

The Firebird boiler used in this study is designed to operate in condensing mode but in practice this depends on a number of different parameters. In order for condensation to occur, the temperature of the flue gas must drop to below the dew point temperature. The dew point temperature is defined as the temperature at which condensation begins when the air is cooled at constant pressure, or in other words the saturation temperature of water corresponding to the vapour pressure [20]. This occurred during start-up when the boiler was getting up to temperature, but once the system had reached steady state, with a water flow temperature of 70-75°C, and flue temperature of over 80°C, the water vapour no longer condensed. Hence, all of the steady state samples reported in this section were performed with the boiler operating in non-condensing mode.

The PM tests performed on the gas boiler turned out as expected. As there are so few impurities in gas, and because it burns so cleanly, the amount of particulate matter produced is generally negligible. That turned out to be the case in the current measuring programme and the emission factor was miniscule, and there was no visible PM on the filter paper after each test. The weighted average emission factor for PM from gas

combustion turned out to be 0.03 g/GJ, a value almost 15 times smaller than the EMEP value, and so low as to make a very small contribution to overall Irish PM levels.

NO_x test results

The NO_x test results tell their own story as well. In contrast to the PM results from the wood-pellet boiler, the NO_x emission factor did not vary much with boiler operating mode. In fact, the measured emission factor remained reasonably constant across the five different power levels, only finding a significantly lower value during the initial start-up stages. At that particular stage of the start-up process, the combustion chamber temperature was still very low, certainly too low for any thermal NO_x to form, and probably too low to release any fuel-bound nitrogen from the fuel. Fuel-NO_x is the dominant formation mechanism for combustion of biomass fuels because the combustion temperature is much lower than for fossil fuel combustion, where thermal NO_x predominates.

The oil- and gas-fired boilers exhibited different trends when it came to their NO_x emission profile. Firstly, even the highest value for each boiler was considerably less than the value published in the EMEP guidebook, following previous trends. For the oil boiler, the steady state emission factor was 42.82 g/GJ, rising slightly to reach a maximum of 48.82 g/GJ during the warm start operating phase. Once again, similar to the wood-pellet boiler, the duty cycle on the oil-fired boiler did not have a significant effect on the quantity of NO_x produced.

The results from the gas-fired boiler showed very low dependence of the NO_x emission factor on the boiler duty cycle as well. The

steady state tests had very repeatable results, with the average emission factor turning out to be 25.81 g/GJ. A minimum value of 16.75 g/GJ was measured during the warm start phase, the reduction as a result of combustion temperatures in the initial phases of the start-up being too low for thermal NO_x to form.

Field testing campaign

As described in the results section, the aim of the field measurement campaign was to compare the NO_x emission factors measured for each of the three boilers in the laboratory to boilers installed in houses and office buildings, to see if there was any major discrepancy between real-world installations and a modern, state-of-the-art unit operating under tightly controlled conditions. The largest deviation was found with the domestic gas boilers with the steady state emission factor of 48.30 g/GJ being almost twice the value of 25.81 g/GJ measured in the laboratory. This was because the boiler used in the laboratory was a brand new boiler with a low NO_x burner, whereas most of the gas boilers sampled were older and didn't have burners with such advanced technology. The commercial gas boilers that were sampled were more modern and representative of modern technology and, as a result, the measured NO_x emission factor was in good agreement with the value found in the laboratory. It is difficult to draw concrete conclusions from the wood boiler samples, as in the case of the commercial scale unit, only one boiler was sampled and it used a different fuel type than the pellet boiler in the laboratory. Of the two domestic boilers sampled, one was the same as the model in the laboratory, and exhibited a similar emission factor, while the other unit

had poorer combustion control and, hence, a significantly higher emission factor. Finally, the results taken from oil boilers sampled in the field were in good agreement with the laboratory results. In the case of the commercial boilers, the two sampled in Monaghan were not operating correctly on the day the test was performed, and their unusually low NO_x emission factor is not representative of a typical boiler. The two units in Mayo were well-serviced and set up correctly and their NO_x levels were almost identical to the domestic oil emission factors and close to the laboratory derived value.

Non-road transport

Over 289,000 flights were analysed to identify suitable aircraft types. These were then grouped into generic groups and analysed using Tier 1, 2 and 3 methodologies. It was found that the largest uncertainty in the estimates related to the quantity of fuel consumed. The main source of data for this was the SEAI energy balance but there was a large discrepancy (70%) between this value and the theoretical value, as calculated using the Tier 3 method. The other variable was the emission factors themselves but these are available from the engine manufacturer data sheets and can be assumed to be accurate. Even allowing for the fuel consumption uncertainty, it was found that the EPA's current inventory report overestimates NO_x LTO emission factors by an average of 30% compared to Tier 2 figures. Larger overestimates occurred with regard to international LTO cycles. For the cruise part of flight activities, the EPA's emission factor of 4.41 g/kg was almost four times lower than the value of 16.77 g/kg found using the Tier 3 method, but that due to discrepancies in the fuel consumption

estimates the total quantity of emissions did not vary too much.

As Ireland is a small country, the contribution of domestic flights to overall emissions is low. An average flight distance of 125 nautical miles was used for the Tier 3 analysis, this being approximately equal to the average of distance between Dublin, Cork and Shannon airports. This simplifying assumption means that other domestic flights having a different journey length to 125 nm will produce different levels of emissions to that predicted by the Tier 3 method. Another aspect of Ireland's domestic flights is that, as a LTO cycle is a standard sequence that every flight must go through, the short 125 nm distance means that the LTO cycle accounts for a far greater proportion of the total emissions than a typical international flight. For example, on a

typical domestic flight, a Boeing B737-400 produces a total NO_x emission of 17.7 kg, with 8.3 kg, or 47% coming during the LTO phases. In contrast, for the same aircraft flying from Dublin to Cairo airport, the total NO_x emissions would be 114.4 kg but the LTO level remains the same at 8.3 kg, just 7% of the total in this case.

For rail transport, the picture is somewhat clearer. The NO_x quantities estimated in the EPA inventory are consistently lower than either the Tier 1 or Tier 2 analysis would suggest. In contrast, however, it was found that the EPA method was considerably overestimating the PM levels. A similar story was uncovered in the marine navigation sector, with NO_x emissions currently being underestimated by over 40%, while PM levels were overestimated by 20% when compared to the Tier 1 method.

9. Conclusions and Recommendations

From the tests and analysis carried out in this research, it is clear that there is significant scope for improving the current emission inventories. The testing programme carried out on the three residential heating boilers consistently found that the measured emission factor was lower, in some cases orders of magnitude lower than the current value listed in the EMEP Guidebook and being used when compiling inventory reports.

The measurement programme sought to investigate what effect a boiler's duty cycle has on the total quantity of emissions. A variety of different installations were monitored to determine a suitable duty cycle. This cycle was then applied to the boilers in the test facility and corresponding NO_x and PM concentrations measured. For PM, no significant dependence was found for the oil and gas boilers, but for the wood-pellet boiler, it was found that the level of PM emitted was very reliant on the operating condition and load on the boiler, for example, an almost five-fold increase in PM emission factors between steady state operation at nominal output and that of cold start-up. For NO_x, a link between boiler operating state and emission factor was found for all three fuel types. The contribution of the various operating states for the three boiler types to the overall running time was examined in detail during the duty cycle data gathering phase, and this was used to derive a 'weighted average' emission factor to be used when compiling emission inventory reports.

An extra aspect that this project has uncovered is the effect of improved boiler technology on emissions concentration. The boilers that were tested in the laboratory were all modern, state-of-the-art units, with two of them being designed for condensing operation. However, a large proportion of boilers in the state is quite old and may not be well-serviced. A field measurement campaign was undertaken to investigate whether older boilers produced higher levels of emissions. In the vast majority of cases, it was found that there was no appreciable difference between the concentration of pollutants in the flue gas from a new or old boiler. There were a couple of exceptions with modern gas boilers that had low-NO_x burners, which lead to significantly lower NO_x levels. However, where new technology has lead to a reduction in emissions is in improved appliance efficiency. As combustion appliances are better designed, and houses now have better insulation levels than they had in the past, the quantity of fuel required to satisfy the heating demand of a building is decreasing. As less fuel is being burned, even if there was no improvement in emission factor, the total quantity of emissions being produced is falling, which brings considerable improvements in air quality.

Recommendations:

Under emission inventory submission guidelines, a member state is permitted to use their own country-specific emission factors instead of the Guidebook values, provided that doing so improves the

accuracy of the inventory. As this study has comprehensively examined the heating sector in Ireland, simulated typical Irish heating loads and accurately measured emission factors under controlled conditions, it is recommended that when the next emission inventory report is being compiled, the emission factors currently in use be replaced by the values derived in this study.

In light of the dramatically lower emission factors measured in this study, it is recommended that the current report be reviewed as soon as possible to take account of the new information that these findings offer. As newer technology condensing boilers, in conjunction with better building standards, are reducing fuel consumption and, hence, emission levels, it is recommended that the current grant scheme available through SEAI for retrofitting houses with improved insulation and heating systems continue, and be expanded where budgetary constraints allow.

As a state-of-the-art emissions measurement facility has been developed in UCD, it should be leveraged into the future. One drawback with the current project was that the impactor used for measuring PM concentration was slow and tricky to use, and an upgraded instrument, such as an electrical low pressure impactor (ELPI), not only could take results much more quickly, but being portable, it could be taken out on a field measuring campaign, something which was not possible with the current level of equipment but which would be very desirable to know. One area that this project has not touched on is the fate of pollutants once they leave the boiler flue and disperse into the atmosphere. An investigation into ambient

NO_x and PM levels would give an excellent insight into the environmental and health impacts of air pollutants.

For the non-road transport sector, there are a number of recommendations to be made. Firstly, in relation to both the marine navigation and rail transport sectors, it was found that total NO_x levels were being underestimated when compared to either the Tier 1 or Tier 2 methods. This should be addressed in the next inventory report to be submitted to CLRTAP. On the other hand, it was found that PM estimates for both of those sectors were being overestimated, so there is scope for improving the emission inventory here. For the air transport sector, there are potential large improvements in NO_x inventories but due to a lack of reliable PM emission factor data, it was not possible to add any further knowledge or insight to the PM inventory. It was found that the current inventory estimates for NO_x from aviation were being overestimated and that this is an area that could be improved on immediately. Another point was that in the 2009 inventory report, the emission factors being used were actually from the 2002 edition of the EMEP guidebook. As a matter of course, this can be upgraded to use the most up-to-date guidebook values. One point which was central to the accuracy of non-road transport emission inventories was that of fuel consumption data. Significant discrepancies were uncovered between the amount of fuel used in the aviation sector and the theoretical amount, as determined by the Tier 3 methodology and backed up when a similar analysis was performed using the ICAO emission factors. Fuel consumption data can also lead to doubts in inventory accuracy for the rail and marine

transport sectors as well and this is definitely an area which could be improved on.

However, it is worth remembering that the contribution of the non-road transport sector to overall emission levels is very small. Aviation accounts for 5% of Ireland's TPER, while rail transport is responsible for a tiny 0.3% of total energy, so any improvement in the data collection process for these sectors

will have a diminishing effect on the overall inventory total. In contrast, the road transport sector is responsible for 23.3% and the residential and commercial heating sector 22.3% of Ireland's total energy consumption. Therefore, it makes sense to focus resources on improving the heating and road transport inventories, as an improvement in the accuracy of data for these sectors can have a dramatic effect on the final result.

References

1. Community, E. (2001). 'Communication from the commission, the Clean Air for Europe (CAFE) Programme: Towards a Thematic strategy for Air Quality'. **COM 245 final**, (4/5/2001).
2. Community, E. (2005). 'Communication from the Commission to the Council and the European parliament. Thematic Strategy on Air Pollution'. **COM 446 final**, (21/09/2005).
3. European Parliament, C. (2001). 'Directive 2001/80/EC of the European Parliament and of the Council on the limitation of the emission of certain pollutants into the air from large combustion plants'. **OJ L 309**.
4. Ireland, G.O. (2007). 'Census 2006 - Volume 6: Housing'. Central Statistics Office (CSO), Cork.
5. Oden, S. (1968). 'The acidification of air precipitation and its consequences in the natural environment'. Statens Naturvetenskaplige Forskningsrid, Stockholm.
6. UNECE. (1979). *Convention on Long Range Transboundary Air Pollution*. Available from: <http://www.unece.org/env/lrtap/full%20text/1979.CLRTAP.e.pdf>, last accessed 15th June 2011.
7. Sliggers, J. and Kakebeeke, W. (2004). 'Clearing the Air: 25 years of the Convention on Long Range and Transboundary Air Pollution'. UNECE, Geneva.
8. EMEP. (2011). *What is EMEP?* Available from: http://www.emep.int/emep_description.html, last accessed 10th March 2011.
9. IIASA. (2004). 'The RAINS review 2004: Modelling health impacts of PM'. International Institute for Applied Systems Analysis, Laxenberg, Austria.
10. Commission, E. (2011). *European Commission web site on Greenhouse Gas and Air Pollution Interaction and Synergies (GAINS)*. Available from: <http://gains.iiasa.ac.at/index.php/home-page>, last accessed 14 March 2011.
11. Community, E. (2001). 'Directive 2001/80/EC of the European Parliament and of the Council on the limitation of the emission of certain pollutants into the air from large combustion plants'. **L309**, (27/11/2001), 1-27.
12. Community, E. (1996). 'Council Directive 96/61/EC concerning integrated pollution prevention and control'. *Official Journal of the European Communities*, **L242**, (10/10/1996).
13. Howley, M., Dennehy, E. and O'Gallachoir, B. (2010). 'Energy in Ireland 1990-2009. 2010 Report'. SEAI, Sustainable Energy Authority of Ireland, Energy Policy Statistical Support Unit.
14. O'Leary, F., Howley, M. and Ó'Gallachóir, B. (2008). 'Energy in the Residential Sector: 2008 Report'. Sustainable Energy Ireland, SEI, Available online at: http://www.seai.ie/News_Events/Press_Releases/Energy_in_the_Residential_Sector_FNL.pdf Last accessed 8/4/2011.

15. SEAI. (2011). *Sustainable Energy Authority of Ireland, Statistics Database*. Available from: <http://www.cso.ie/px/sei/database/sei/sei.asp>, last accessed 10th March 2011.
16. Dimplex, G. (2005). 'The European Heating Products Market 2005 update: Ireland'. ConsultGB, Canterbury.
17. Gais, B. (2009). 'Annual Report and Financial Statements 2009'. Bord Gais, Eireann.
18. Stern, A.C., Boubel, R.W., Turner, D.B. and Fox, D.L. (1984). *Fundamentals of Air Pollution*. Academic press, Inc, London, Second Edition.
19. Turns, S.R. (1996). *An Introduction to Combustion: Concepts and Applications*. McGraw-Hill.
20. Cengel, Y.A. and Boles, M.A. (2007). *Thermodynamics: An Engineering Approach*. McGraw-Hill, Sixth Edition (S.I. Units).
21. Singer, J.G. (1981). *Combustion: Fossil Power Systems*. Combustion Engineering, INC.,
22. Heywood, J.B. (1988). *Internal Combustion Engine Fundamentals*. McGraw-Hill, International Edition.
23. Ledbetter, J.A. (1972). *Air Pollution Part A: Analysis*. Marcel Dekker, Inc, New York,
24. ManageMyHome.com (2008). Available from: www.managemyhome.com/mmh/ideas/ViewCategoryNavigationAction.action?level1=Topics&level2Index=10&level3Index=8&uid=37f8c4f4-e35f-11db-8a44-b57745f8a18d&contentType
<<http://www.managemyhome.com/mmh/ideas/ViewCategoryNavigationAction.action?level1=Topics&level2Index=10&level3Index=8&uid=37f8c4f4-e35f-11db-8a44-b57745f8a18d&contentType>>, last accessed June 10th 2008.
25. Wagner, A.Y., Livbjerg, H., Kristensen, P.G. and Glarborg, P. (2010). 'Particle Emissions from Domestic Gas Cookers'. *Combustion Science and Technology*, **182**, 10, 1511 - 1527.
26. Pye, S., Thistlethwaite, G., Adams, M., Woodfield, M., Goodwin, J., Forster, D. and Holland, M. (2004). 'Costs and environmental effectiveness of options for reducing air pollution from small-scale combustion installations. Final report to European Commission DG Environment.' AEA Technology Environment, Harwell, UK.
27. Li, Y. and Liu, H. (2000). 'High-pressure densification of wood residues to form an upgraded fuel'. *Biomass and Bioenergy*, **19**, 3, 177-186.
28. Bergström, D., Israelsson, S., Öhman, M., Dahlqvist, S.-A., Gref, R., Boman, C. and Wästerlund, I. (2008). 'Effects of raw material particle size distribution on the characteristics of Scots pine sawdust fuel pellets'. *Fuel Processing Technology*, **89**, 12, 1324-1329.
29. CEN. (2005). *CEN/TS 14961: Solid biofuels — Fuel specifications and classes*. European Committee for Standardisation, Brussels.
30. Fiedler, F. (2004). 'The state of the art of small-scale pellet-based heating systems and relevant regulations in Sweden, Austria and Germany'. *Renewable and Sustainable Energy Reviews*, **8**, 3, 201-221.

31. Che, D., Liu, Y. and Gao, C. (2004). 'Evaluation of retrofitting a conventional natural gas fired boiler into a condensing boiler'. *Energy Conversion and Management*, **45**, 20, 3251-3266.
32. Government, I. (2007). 'Building Regulations Part L'. **S.I. No. 854**,
33. Grohn, A., Suonmaa, V., Auvinen, A., Lehtinen, K.E.J. and Jokiniemi, J. (2009). 'Reduction of Fine Particle Emissions from Wood Combustion with Optimized Condensing Heat Exchangers'. *Environmental Science & Technology*, **43**, 16, 6269-6274.
34. Van Kemenade, H.P. *Particulate emission reduction in small-scale biomass combustion plants by a condensing heat exchanger*, in *5th European Thermal-Sciences Conference*. 2008: The Netherlands.
35. Nussbaumer, T. (2003). 'Combustion and Co-combustion of Biomass: Fundamentals, Technologies, and Primary Measures for Emission Reduction'. *Energy & Fuels*, **17**, 6, 1510-1521.
36. Hinds, W.C. (1999). *Aerosol Technology: Properties, Behaviour, and Measurement of Airborne Particles*. John Wiley & Sons, Second Edition.
37. Lighty, J.S., Veranth, J.M. and Sarofim, A.F. (2000). 'Combustion aerosols: factors governing their size and composition and implications to human health'. *Journal of the Air & Waste Management Association*, **50**, 9, 1565-1618.
38. Krombach, F., Münzing, S., Allmeling, A.-M.J., Gerlach, T., Behr, J. and Dörger, M. (1997). 'Cell Size of Alveolar Macrophages: An Interspecies Comparison'. *Environmental Health Perspectives*, **105**, Supplement 5, 1261-1263.
39. Stern, A.C. (1977). *Air Pollution Volume II: The Effects of Air Pollution*. Academic Press, Inc, New York, Third Edition
40. Berglund, M., Bostrom, C.E., Bylin, G., Ewetz, L., Gustafsson, L., Moldens, P., Pershagen, G. and Victorin, K. (1993). 'Health Risk evaluation of nitrogen oxides'. *Scandinavian Journal of Work, Environment and Health*, **19**, Suppl. 2, 1-70.
41. Chitano, P., Hosselet, J., Mapp, C. and Fabbri, L. (1995). 'Effect of oxidant air pollutants on the respiratory system: insights from experimental animal research'. *European Respiratory Journal*, **8**, 8, 1357-1371.
42. Kraft, M., Eikmann, T., Kappos, A., Künzli, N., Rapp, R., Schneider, K., Seitz, H., Voss, J.-U. and Wichmann, H.E. (2005). 'The German view: Effects of nitrogen dioxide on human health - derivation of health-related short-term and long-term values'. *International Journal of Hygiene and Environmental Health*, **208**, 4, 305-318.
43. Gauderman, W.J., Avol, E., Gilliland, F., Vora, H., Thomas, D., Berhane, K., McConnell, R., Kuenzli, N., Lurmann, F., Rappaport, E., Margolis, H., Bates, D. and Peters, J. (2004). 'The Effect of Air Pollution on Lung Development from 10 to 18 Years of Age'. *New England Journal of Medicine*, **351**, 11, 1057-1067.
44. WHO. (2003). 'Health Aspects of Air Pollution with Particulate Matter, Ozone and Nitrogen Dioxide'. EUR/03/5042688, World Health Organisation, Bonn, Germany.

45. Naess, O., Nafstad, P., Aamodt, G., Claussen, B. and Rosland, P. (2007). 'Relation between Concentration of Air Pollution and Cause-Specific Mortality: Four-Year Exposures to Nitrogen Dioxide and Particulate Matter Pollutants in 470 Neighborhoods in Oslo, Norway'. *American Journal of Epidemiology*, **165**, 4, 435-443.
46. Rossi, G., Vigotti, M.A., Zanobetti, A., Repetto, F., Gianelle, V. and Schwartz, J. (1999). 'Air Pollution and Cause-Specific Mortality in Milan, Italy, 1980-1989'. *Archives of Environmental Health*, **54**, 3, 158-164.
47. Sunyer, J., Castellsague, J., Saez, M., Tobias, A. and Anto, J.M. (1996). 'Air pollution and mortality in Barcelona'. *Journal of Epidemiology and Community Health*, **50 (Supp 1)**, S76-S80.
48. Evelyn, J. (1661). *Fumifugium or the Inconvenience of the Aer, and Smoake of London Dissipated Together With some Remedies humbly Proposed*. W. Godbid, London.
49. Davis, D.L., Bell, M.L. and Fletcher, T. (2002). 'A Look Back at the London Smog of 1952 and the Half Century Since'. *Environmental Health Perspectives*, **110**, 12, A734.
50. Community, E. (1999). 'Council Directive 1999/30/EC of 22 April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air'. **L163/41**, (26 June 1999), 1-20.
51. Harrison, R.M. and Yin, J. (2000). 'Particulate matter in the atmosphere: which particle properties are important for its effects on health?'. *The Science of The Total Environment*, **249**, 1-3, 85-101.
52. Dockery, D.W., Pope, C.A., Xu, X., Spengler, J.D., Ware, J.H., Fay, M.E., Ferris, B.G. and Speizer, F.E. (1993). 'An Association between Air Pollution and Mortality in Six U.S. Cities'. *N Engl J Med*, **329**, 24, 1753-1759.
53. Pope, C., 3rd, Thun, M., Namboodiri, M., Dockery, D., Evans, J., Speizer, F. and Heath, C., Jr. (1995). 'Particulate air pollution as a predictor of mortality in a prospective study of U.S. adults'. *Am. J. Respir. Crit. Care Med.*, **151**, 3, 669-674.
54. Bolling, A.K., Pagels, J., Yttri, K.E., Barregard, L., Sallsten, G., Schwarze, P.E. and Boman, C. (2009). 'Health effects of residential wood smoke particles: the importance of combustion conditions and physicochemical particle properties'. *Particle and Fibre Toxicology*, **6**, 29,
55. Ostro, B., Broadwin, R., Green, S., Feng, W.-Y. and Lipsett, M. (2006). 'Fine particulate air pollution and mortality in nine California counties: results from CALFINE'. *Environmental Health Perspectives*, **114**, 29-33.
56. Pope, C.A., Burnett, R.T., Thun, M.J., Calle, E.E., Krewski, D., Ito, K. and Thurston, G.D. (2002). 'Lung Cancer, Cardiopulmonary Mortality, and Long-term Exposure to Fine Particulate Air Pollution'. *JAMA: The Journal of the American Medical Association*, **287**, 9, 1132-1141.
57. Pope, C.A. and Dockery, D.W. (2006). 'Health effects of fine particulate air pollution: lines that connect.'. *Journal of Air and Waste Management*, **56**, 709-742.
58. Kennedy, I.M. (2007). 'The health effects of combustion-generated aerosols'. *Proceedings of the Combustion Institute*, **31**, 2, 2757-2770.

59. Maynard, A.D. and Maynard, R.L. (2002). 'A derived association between ambient aerosol surface area and excess mortality using historic time series data'. *Atmospheric Environment*, **36**, 36-37, 5561-5567.
60. Dockery, D.W. and Stone, P.H. (2007). 'Cardiovascular Risks from Fine Particulate Air Pollution'. *New England Journal of Medicine*, **356**, 5, 511-513.
61. Samet, J.M., Dominici, F., Curriero, F.C., Coursac, I. and Zeger, S.L. (2000). 'Fine Particulate Air Pollution and Mortality in 20 U.S. Cities, 1987-1994'. *New England Journal of Medicine*, **343**, 24, 1742-1749.
62. Neas, L.M. (2000). 'Fine particulate matter and cardiovascular disease'. *Fuel Processing Technology*, **65-66**, 55-67.
63. Flanagan, P.J. (1986). 'Air Quality in Ireland: the present position'. An Foras Forbartha, Dublin.
64. Air Pollution Act, 1987 (Marketing, Sale and Distribution of Fuels) Regulations, 1990, SI 123 of 1990, Dublin:Government Publications Office
65. Clancy, L., Goodman, P., Sinclair, H. and Dockery, D.W. (2002). 'Effect of air-pollution control on death rates in Dublin, Ireland: an intervention study'. *The Lancet*, **360**, 9341, 1210-1214.
66. Brook, R.D., Franklin, B., Cascio, W., Hong, Y., Howard, G., Lipsett, M., Luepker, R., Mittleman, M., Samet, J., Smith, S.C., Jr. and Tager, I. (2004). 'Air Pollution and Cardiovascular Disease: A Statement for Healthcare Professionals From the Expert Panel on Population and Prevention Science of the American Heart Association'. *Circulation*, **109**, 21, 2655-2671.
67. Kreyling, W., Semmler-Behnke, M. and Möller, W. (2006). 'Health implications of nanoparticles'. *Journal of Nanoparticle Research*, **8**, 5, 543-562.
68. Burchiel, S.W., Lauer, F.T., Dunaway, S.L., Zawadzki, J., McDonald, J.D. and Reed, M.D. (2005). 'Hardwood smoke alters murine splenic T cell responses to mitogens following a 6-month whole body inhalation exposure'. *Toxicology and Applied Pharmacology*, **202**, 3, 229-236.
69. Seagrave, J., McDonald, J.D., Reed, M.D., Seilkop, S.K. and Mauderly, J.L. (2005). 'Responses to Subchronic Inhalation of Low Concentrations of Diesel Exhaust and Hardwood Smoke Measured in Rat Bronchoalveolar Lavage Fluid'. *Inhalation Toxicology*, **17**, 12, 657-670.
70. Barrett, E.G., Henson, R.D., Seilkop, S.K., McDonald, J.D. and Reed, M.D. (2006). 'Effects of Hardwood Smoke Exposure on Allergic Airway Inflammation in Mice'. *Inhalation Toxicology*, **18**, 1, 33-43.
71. Reed, M.D., Campen, M.J., Gigliotti, A.P., Harrod, K.S., McDonald, J.D., Seagrave, J.C., Mauderly, J.L. and Seilkop, S.K. (2006). 'Health Effects of Subchronic Exposure to Environmental Levels of Hardwood Smoke'. *Inhalation Toxicology*, **18**, 8, 523-539.
72. Samuelson, M., Nygaard, U.C. and Løvik, M. (2008). 'Allergy adjuvant effect of particles from wood smoke and road traffic'. *Toxicology*, **246**, 2-3, 124-131.

73. Tesfaigzi, Y., Singh, S.P., Foster, J.E., Kubatko, J., Barr, E.B., Fine, P.M., McDonald, J.D., Hahn, F.F. and Mauderly, J.L. (2002). 'Health Effects of Subchronic Exposure to Low Levels of Wood Smoke in Rats'. *Toxicological Sciences*, **65**, 1, 115-125.
74. Tesfaigzi, Y., McDonald, J.D., Reed, M.D., Singh, S.P., De Sanctis, G.T., Eynott, P.R., Hahn, F.F., Campen, M.J. and Mauderly, J.L. (2005). 'Low-Level Subchronic Exposure to Wood Smoke Exacerbates Inflammatory Responses in Allergic Rats'. *Toxicological Sciences*, **88**, 2, 505-513.
75. Nel, A.E., Diaz-Sanchez, D. and Ning, L. (2001). 'The role of particulate pollutants in pulmonary inflammation and asthma: evidence for the involvement of organic chemicals and oxidative stress.' *Current Opinion in Pulmonary Medicine*, **7**, 1, 20-26.
76. Ghio, A.J., Kim, C. and Devlin, R.B. (2000). 'Concentrated Ambient Air Particles Induce Mild Pulmonary Inflammation in Healthy Human Volunteers'. *Am. J. Respir. Crit. Care Med.*, **162**, 3, 981-988.
77. Junker, R., Heinrich, J., Ulbrich, H., Schulte, H., Schonfeld, R., Kohler, E. and Assmann, G. (1998). 'Relationship Between Plasma Viscosity and the Severity of Coronary Heart Disease'. *Arteriosclerosis, Thrombosis and Vascular Biology*, **18**, 6, 870-875.
78. Jokiniemi, J.K., Hytonen, K., Tissari, J., Obernberger, I., Brunner, T., Barnthaler, G., Friesenbichler, J., Salonen, R.O., Hirvonen, M.-R., Jalava, P., Pennanen, A., Happonen, M., Vallius, M., Markkanen, P., Hartmann, H., Turowski, P., Rossmann, P., Ellner-Schubert, F., Boman, C., Pettersson, E., Wiinikka, H., Hillamo, R., Saarnio, K., Frey, A., Saarikoski, S., Timonen, H., Teinila, K., Aurela, M., Sillanpää, M., Bellmann, B., Sandstrom, T., Sehlstedt, M. and Forsberg, B. (2008). '*Biomass combustion in residential heating: Particulate measurements, sampling, and physicochemical and toxicological characterisation*. Final report of the project 'Biomass-PM' funded by ERA-NET Bioenergy Programme 2007-2008'. Report 1/2008, University of Kuopio, Kuopio, Finland.
79. Karlsson, H.L., Ljungman, A.G., Lindbom, J. and Möller, L. (2006). 'Comparison of genotoxic and inflammatory effects of particles generated by wood combustion, a road simulator and collected from street and subway'. *Toxicology Letters*, **165**, 3, 203-211.
80. Leonard, S.S., Wang, S., Shi, X., Jordan, B.S., Castranova, V. and Dubick, M.A. (2000). 'Wood smoke particles generate free radicals and cause lipid peroxidation, DNA damage, NF[κ]B activation and TNF-[α] release in macrophages'. *Toxicology*, **150**, 1-3, 147-157.
81. Kocbach, A., Namork, E. and Schwarze, P.E. (2008). 'Pro-inflammatory potential of wood smoke and traffic-derived particles in a monocytic cell line'. *Toxicology*, **247**, 2-3, 123-132.
82. Kocbach, A., Herseth, J.I., Låg, M., Refsnes, M. and Schwarze, P.E. (2008). 'Particles from wood smoke and traffic induce differential pro-inflammatory response patterns in co-cultures'. *Toxicology and Applied Pharmacology*, **232**, 2, 317-326.
83. Brown, D.M., Stone, V., Findlay, P., MacNee, W. and Donaldson, K. (2000). 'Increased inflammation and intracellular calcium caused by ultrafine carbon black is independent of transition metals or other soluble components'. *Occupational and Environmental Medicine*, **57**, 10, 685-691.

84. Höhr, D., Steinfartz, Y., Schins, R.P.F., Knaapen, A.M., Martra, G., Fubini, B. and Borm, P.J.A. (2002). 'The surface area rather than the surface coating determines the acute inflammatory response after instillation of fine and ultrafine TiO₂ in the rat'. *International Journal of Hygiene and Environmental Health*, **205**, 3, 239-244.
85. Monteiller, C., Tran, L., MacNee, W., Faux, S., Jones, A., Miller, B. and Donaldson, K. (2007). 'The pro-inflammatory effects of low-toxicity low-solubility particles, nanoparticles and fine particles, on epithelial cells in vitro: the role of surface area'. *Occupational and Environmental Medicine*, **64**, 9, 609-615.
86. Murphy, S.A., Berube, K.A. and Richards, R.J. (1999). 'Bioreactivity of carbon black and diesel exhaust particles to primary Clara and type II epithelial cell cultures'. *Occupational and Environmental Medicine*, **56**, 12, 813-819.
87. Anselm, A., Heibel, T., Gebhart, J. and Ferron, G. (1990). "'In vivo"-studies of growth factors of sodium chloride particles in the human respiratory tract'. *Journal of Aerosol Science*, **21**, Supplement 1, S427-S430.
88. Dautrebande, L. and Walkenhorst, W. (1961). *Inhaled Particles and Vapours*. Pergamon, Oxford, England,
89. Londahl, J., Pagels, J., Boman, C., Swietlicki, E., Massling, A., Rissler, J., Blomberg, A., Bohgard, M. and Sandstrom, T. (2008). *Deposition of Biomass Combustion Aerosol Particles in the Human Respiratory Tract*, in *Inhalation Toxicology*. Taylor & Francis Ltd. p. 923-933.
90. Wiebert, P., Sanchez-Crespo, A., Falk, R., Philipson, K., Lundin, A., Larsson, S., MÅller, W., Kreyling, W. and Svartengren, M. (2006). *No Significant Translocation of Inhaled 35-nm Carbon Particles to the Circulation in Humans*, in *Inhalation Toxicology*. Taylor & Francis Ltd. p. 741-747.
91. De Nevers, N. (1995). *Air Pollution Control Engineering*. McGraw-Hill,
92. Karim, M.R., Sekine, M. and Ukita, M. (2002). 'Simulation of eutrophication and associated occurrence of hypoxic and anoxic condition in a coastal bay in Japan'. *Marine Pollution Bulletin*, **45**, 1-12, 280-285.
93. Faith, W.L. and Atkisson, A.A. (1972). *Air Pollution*. Wiley-Interscience, Second Edition.
94. Grantz, D.A., Garner, J.H.B. and Johnson, D.W. (2003). 'Ecological effects of particulate matter'. *Environment International*, **29**, 2-3, 213-239.
95. Farmer, A.M. (1993). 'The effects of dust on vegetation--a review'. *Environmental Pollution*, **79**, 1, 63-75.
96. CEN. (2007). *EN 15270:2007. Pellet burners for small heating boilers - Definitions, requirements, testing, marking*. European Committee for Standardisation, Brussels.
97. CEN. (1999). *EN 303-4:1999. Heating boilers - Part 4: Heating boilers with forced draught burners - Special requirements for boilers with forced draught oil burners with outputs up to 70 kW and a maximum operating pressure of 3 bar - Terminology, special requirements, testing and marking*. European Committee for Standardisation, Brussels.
98. CEN. (1999). *EN 303-5:1999. Heating boilers - Part 5: Heating boilers for solid fuels, hand and automatically fired, nominal heat output of up to 300 kW - Terminology, requirements, testing and marking*. European Committee for Standardisation, Brussels.

99. CEN. (2006). *EN 14785:2006. Residential space heating appliances fired by wood pellets - Requirements and test methods*. European Committee for Standardisation, Brussels.
100. CEN. (2006). *prEN 15502-1:2006. Gas-fired central heating boilers - Part 1: General requirements and tests*. European Committee for Standardisation, Brussels.
101. Communities, E. (2001). 'Directive 2001/81/EC of the European Parliament and of the Council of 23 October 2001 on national emission ceilings for certain atmospheric pollutants'. **L309**, (27/11/2001), 1-14.
102. EEA. (2011). 'NEC Directive Status Report 2010: Reporting by the Member States under Directive 2001/81/EC of the European Parliament and of the Council of 23 October 2001 on national emission ceilings for certain atmospheric pollutants'. EEA Technical Report No. 3/2011, European Environment Agency, Copenhagen.
103. Communities, E. (2000). 'Directive 2000/69/EC of the European Parliament and of the Council of 16 November 2000 relating to limit values for benzene and carbon monoxide in ambient air'. **L313/12**, (13/12/2000), 1-10.
104. Communities, E. (2008). 'Directive 2008/50/EC of the European parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe'. *Official Journal of the European Communities*, **L152**, (11/6/2008), 1-44.
105. Environmental Protection Agency Act. (1992). SI 7 of 1992, Ireland: Government Publications Office.
106. EPA. (2011). Environmental Protection Agency, Available from: <http://www.epa.ie/whatwedo/>, last accessed 31st March 2011.
107. McGettigan, M., Duffy, P. and Hyde, B. (2009). 'Ireland: Informative Inventory Report 2009'. Environmental Protection Agency (EPA), Dublin.
108. Johansson, L.S., Leckner, B., Gustavsson, L., Cooper, D., Tullin, C. and Potter, A. (2004). 'Emission characteristics of modern and old-type residential boilers fired with wood logs and wood pellets'. *Atmospheric Environment*, **38**, 25, 4183-4195.
109. Brunner, T., Barnthaler, G. and Obernberger, I. (2008). 'Evaluation of parameters determining PM emissions and their chemical composition in modern residential biomass heating appliances'. *In Proceedings of World Bioenergy 2008*, Jonkoping, Sweden, 27 - 29 May 2008, pp.
110. Wiinikka, H. and Gebart, R. (2004). 'Experimental investigations of the influence from different operating conditions on the particle emissions from a small-scale pellets combustor'. *Biomass and Bioenergy*, **27**, 6, 645-652.
111. Hays, M.D., Beck, L., Barfield, P., Lavrich, R.J., Dong, Y. and Vander Wal, R.L. (2008). 'Physical and Chemical Characterization of Residential Oil Boiler Emissions'. *Environmental Science & Technology*, **42**, 7, 2496-2502.
112. Agency, M.E.P. (2008). 'Regulations for the Prevention of Air Pollution from Ships'. *International Maritime Organisation*,
113. NI. (2011). National Instruments, Available from: <http://zone.ni.com/devzone/cda/tut/p/id/4644>, last accessed 14th March 2011.
114. Testo. (2010). Testo Ltd., Available from: <http://www.testo350.com/>, last accessed 14th March 2011.

115. Standardization, I.O.f. (2003). *BS ISO 9096:2003. Stationary source emissions — Manual determination of mass concentration of particulate matter*. ISO,
116. USEPA Method 5 - *Determination of particulate matter emissions from stationary sources*. USEPA. Available from: <http://www.epa.gov/ttn/emc/promgate/m-05.pdf>, last accessed 19th July 2011.
117. Nussbaumer, T., Klippel, N. and Johansson, L.S. (2008). 'Survey on measurements and emission factors on particulate matter from biomass combustion in IEA countries'. In *Proceedings of 16th European Biomass Conference and Exhibition*, Valencia, Spain, 2nd-6th June 2008, pp. 1-14.
118. Nussbaumer, T. (2010). 'Overview on Technologies for Biomass Combustion and Emission levels of Particulate Matter'. ISBN 3-908705-21-5, Swiss Federal Office for the Environment (FOEN) as a contribution to the Expert Group on Techno-Economic Issues (EGTEI) under the Convention on Long-Range Transboundary Air Pollution, Verenum Press Zürich (Switzerland), 7 May 2010, Zurich, CH.
119. Dekati. (2010). *DLPI User Manual Version 3.62*, Ltd, D., Tampere, Finland. Available from: <http://dekati.com/cms/system/files/DLPI+3.62.pdf>, last accessed: 17th August 2011.
120. Pall. (2002). *Pall Life Sciences, Laboratory Filtration and Separation Catalog. The ultimate filtration, purification, and separation reference*.
121. SEI. (2006). *Sustainable Energy Ireland, Launch of Greener Homes Scheme. Press Release*. Available from: http://www.seai.ie/News_Events/Press_Releases/2006/, last accessed 8/4/2011.
122. KWB. (2011). *KWB Easyfire Pellet Boilers*. Available from: http://www.kwb.at/at/index.php?option=com_content&task=view&id=192&Itemid=170, last accessed 8/4/2011.
123. ETA. (2011). *ETA Heiztechnik GmbH PE Pellet Boilers*. Available from: <http://www.eta.co.at/index.php?id=18&L=0>, last accessed 8/4/2011.
124. Bretby, T. (2010). 'Test Report - Balcas Timber Ltd'.
125. Phyllis. (2010). *Phyllis, database for biomass and waste*. Energy Research Centre of the Netherlands Available from: <http://www.ecn.nl/phyllis>, last accessed 15th April 2011.
126. Brunner T. B.G., Obernberger I. (2008). 'Evaluation of parameters determining PM emissions and their chemical composition in modern residential biomass heating appliances'. In *Proceedings of World Bioenergy 2008*, Jonkoping, Sweden, 27 - 29 May 2008, pp.
127. Caserini, S., Livio, S., Giugliano, M., Grosso, M. and Rigamonti, L. (2010). 'LCA of domestic and centralised biomass combustion: The case of Lombardy (Italy)'. *Biomass and Bioenergy*, **34**, 474-482.
128. EMEP-EEA. (2009) *Air Pollutant Emission Inventory Guidebook - 2009*.
129. Hueglin, C., Gaegauf, C., Kunzel, S. and Burtscher, H. (1997). 'Characterization of Wood Combustion Particles: Morphology, Mobility, and Photoelectric Activity'. *Environmental Science & Technology*, **31**, 12, 3439-3447.

130. Wiinikka, H. (2005). *High Temperature Aerosol Formation and Emission Minimisation during Combustion of Wood Pellets*. Department of Applied Physics and Mechanical Engineering, Lulea University of Technology, Lulea, Sweden.
131. Jiménez, S. and Ballester, J. (2004). 'Formation and Emission of Submicron Particles in Pulverized Olive Residue (Orujillo) Combustion'. *Aerosol Science and Technology*, **38**, 7, 707 - 723.
132. EEA. (2014). 'NEC Directive Status Report 2013: Reporting by the Member States under Directive 2001/81/EC of the European Parliament and of the Council of 23 October 2001 on national emission ceilings for certain atmospheric pollutants'. EEA Technical Report No. 10/2014, European Environment Agency, Copenhagen.
133. Morrin, S. (2011). *Quantifying Particulate Matter and NO_x emission factors for Irish domestic heating appliances*. Ph.D Thesis, School of Electrical, Electronic, and Mechanical Engineering, University College Dublin, Ireland.

List of Acronyms

AFR	Air:Fuel Ratio
CAFE	Clean Air For Europe
CFC	Chlorofluorocarbon
CLRTAP	Convention on Long Range Transboundary Air Pollution
CO	Carbon Monoxide
CO ₂	Carbon Dioxide
DAQ	Data Acquisition System
DLPI	Dekati Low Pressure Impactor
EC	European Community
EDX	Energy Dispersive X-ray Spectroscopy
EEA	European Environment Agency
EF	Emission Factor
ELPI	Electrical Low Pressure Impactor
EMEP	Cooperative Programme for Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe
EPA	Environmental Protection Agency
EU	European Union
FAR	Fuel:Air Ratio
GJ	Gigajoule (10 ⁹ J)
H ₂	Hydrogen
H ₂ O	Water Vapour
ICAO	International Civil Aviation Authority
IIR	Informative Inventory Report
IMO	International Maritime Organisation
LCPD	Large Combustion Plant Directive
LTO	Landing and Takeoff
MTOE	Million Tonnes of Oil Equivalent
NDIR	Non-Dispersive Infra-Red
NECD	National Emissions Ceiling Directive
NH ₃	Ammonia
NMVOG	Non-Methane Volatile Organic Compound
NO	Nitric Oxide
NO ₂	Nitrogen Dioxide
NO _x	Nitrogen Oxide
OCLR	Office of Climate, Licensing and Resource Use
PAH	Polycyclic Aromatic Hydrocarbon
PL	Power Level
PM	Particulate Matter
PTFE	Polytetrafluoroethylene

Improved Emissions Inventories for NO_x and Particulate Matter from Transport and Small Scale Combustion Installations in Ireland (ETASCI)

RH	Relative Humidity
ROS	Reactive Oxygen Species
RTD	Resistance Temperature Detector
SCI	Small Combustion Installation
SEAI	Sustainable Energy Authority of Ireland
SEM	Scanning Electron Microscopy
SO ₂	Sulphur Dioxide
TFIAM	Task Force on Integrated Assessment and Modelling
TFMM	Task Force on Measurements and Modelling
TPER	Total Primary Energy Requirement
UCD	University College Dublin
UHC	Unburnt Hydrocarbons
US	United States
USB	Universal Serial Bus
USEPA	United States Environmental Protection Agency
UV	Ultra Violet
VI	Virtual Instrument
VOC	Volatile Organic Compounds
WHO	World Health Organisation

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Combustion-related emissions of NO_x and particulate matter (PM) have been shown to exert a strongly negative impact on the environment and on human health. As part of its contribution to international mitigation efforts, Ireland is committed to quantifying and reporting national emissions of these pollutants on an annual basis. This project provides improved estimates of Irish emissions from small, distributed, combustion sources, specifically from the Road Transport sector (Work Package 1), and from Non-road Transport, and stationary combustion installations such as domestic boilers (Work Package 2).

Identifying Pressures

Small combustion installations, whether stationary or mobile, emit pollutants near ground level, and in close proximity to humans. It is therefore vital that the levels of emission from these sources are accurately quantified: a difficult challenge. The emissions are estimated by combining activity data – which details fuel use by type and sector – with emission factors (EFs), which quantify the mass of each pollutant emitted per unit of fuel consumed. Whereas accurate activity data is, in general, available, emission factors for many of the fuel and appliance combinations commonly used in Ireland are highly uncertain.

Informing Policy

This project can inform policy in relation to emissions from both stationary, and mobile, combustion sources. With regard to small, stationary sources, emission factors specific to Irish conditions have been developed within the project for some combinations of fuel and appliance. These emission factors should be implemented in future emission inventory estimates, and corresponding emission factors for solid-fuel appliances should be developed as a matter of urgency. With respect to road transport, the project has demonstrated that an inherently high level of uncertainty is embedded in the NO_x emission factors associated with current vehicle technologies. Policymakers should therefore be wary of ascribing too much weight to these data when formulating policy.

Developing Solutions

The Ireland-specific emission factors determined in this project are an intrinsically valuable output; the facilities and expertise developed within the project, however, will be of even greater benefit. That new capacity is currently being used to develop emission factors for small appliances burning a range of solid fuels. This important, but very challenging, task will also identify fuel-appliance combinations that are particularly injurious to the environment or to human health, and if necessary could inform the development of policies that might limit or proscribe their use.

** The ETASCI project is divided into two complementary but distinct work packages (WPs) and final reports. WP1 focuses on emissions from road transport and is detailed in EPA Research Report 148. WP2 examines emissions from non-road transport and from small combustion installations (SCIs) and is detailed in EPA Research Report 149.*

