

Emissions from cofiring coal, biomass and sewage sludge

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Abstract

The practice of cofiring biomass in full-scale coal utility plants is increasing, due to the benefits of reduced fossil fuel based CO₂ emissions. Biomass also tends to have a lower sulphur content than coal and therefore emissions of SO₂ can be reduced. The same is true for NO_x emissions from lower fuel nitrogen content. Further, the lower flame temperatures and different combustion stoichiometry of biomass systems can also result in lower thermal NO_x production. A reduction in ash, especially when cofiring wood, is another advantage. Increased chlorine and/or changes in ash particle adsorbency can help reduce trace element emissions such as mercury and arsenic. However, some biomass materials, such as straw and grass, can have higher potassium and chlorine than coal which may lead to problems such as slagging and fouling. There are also potential issues with respect to changes in the operation of pollution control technologies. For example:

- phosphorus in sewage sludge can react with lime to reduce sulphur capture in fluidised bed systems;
- sewage sludge tends to have higher concentrations of several trace elements such as Cr, Cu, Ni, Pb, Zn and Fe. It has been shown that these elements tend to end up in the fly ash and it is important to ensure that this does not result in limitations on the intended use of such ash.

In most cases, the balance between the characteristics of the coal and biomass and the plant operation can control any negative plant effects. In practice, full-scale coal-fired plants such as Drax in the UK and Fiume Santo in Sardinia note little or no detrimental change in trace element emissions following the introduction of biomass as a co-fuel.

It would appear that, for the most part, the benefits of cofiring biomass far outweigh any negative effects. In fact, it would seem that the majority of environmental impact assessments regard the production, transport and preparation of the biomass fuels as more important than changes in the stack emissions from the plant as a result of the cofiring. Detrimental effects, however, can be an issue for ash management.

Acronyms and abbreviations

ASTM	American Standards for Testing and Materials
BFBC	bubbling fluidised bed combustion
CFBC	circulating fluidised bed combustion
DDGS	dried distilled grains with solubles
EC	European Commission
EDF	Electricité de France
EL	emission limit
EU	European Union
ESP	electrostatic precipitator
FBC	fluidised bed combustion
FGD	flue gas desulphurisation
FIP	feed-in premium
FIT	feed-in tariff
IEA	International Energy Agency
IEA CCC	IEA Clean Coal Centre
IPC	Integrated Pollution Control, UK
LCPD	Large Combustion Plant Directive, EU
MBM	meat and bone meal
MMBtu	million British thermal units
MSW	municipal solid waste
Mtoe	megatonne of oil equivalent
MWe	megawatt electric
MWth	megawatt thermal
NO _x	nitrogen oxides (NO and NO ₂)
OHM	Ontario Hydro Method
PAH	polycyclic aromatic hydrocarbons
PCDD	polychlorinated dibenzo dioxin
PCDF	polychlorinated dibenzo furan
PFBC	pressurised fluidised bed combustion
PKS	palm kernel shells
RDF	refuse derived fuel
ROC	Renewable Obligation Certificate, UK
SCEM	PS® Analytical continuous emissions monitoring system
SCR	selective catalytic reduction
SFBC	swirling fluidised bed combustion
TOC	total organic carbon
WDF	waste derived fuel
WID	Waste Incineration Directive, EU

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I Introduction

The use of biomass as a co-fuel for coal is increasing as biomass may be considered to be carbon-neutral in some situations. This means that less coal needs to be burned for the same amount of electricity produced and, at the same time, the CO₂ emissions are reduced. According to Koppejan and Baxter (2005), replacing 5% of coal with biomass on an energy basis worldwide would produce 40 GWe of power and an emission reduction of 300 Mt/y CO₂. The issue of CO₂ emissions from biomass cofiring is beyond the scope of this report. However, CO₂ reduction is certainly the major motivation behind the move towards biomass cofiring at the plants where this is happening.

In 2004, cofiring of biomass with coal was employed at 135 plants worldwide with most of the plants located in the USA, Germany and Finland (Leckner, 2007). Incentives have been initiated in several countries to promote the use of biomass for energy production. Europe and the USA have set challenging targets for the use of renewable energy sources with biomass combustion representing one of the most promising options.

Biomass materials are more physically and chemically variable than coal and therefore they pose new challenges with respect to maintaining the efficiency of combustion and operation of existing coal-fired plants. Further, the elemental content of biomass materials can be quite different from that of coal and could, in certain situations, lead to increased emissions of certain species. Biomass burning alone is known to be a significant source of emissions, especially particulates, in older combustion systems. However, the cofiring of biomass in new, more efficient boilers or with coal offers the potential for cleaner combustion and lower emissions, if the combustion conditions are optimised. This report concentrates on the effects of cofiring biomass in existing systems that were originally designed for coal combustion alone.

Several previous reports from the IEA Clean Coal Centre (CCC) have concentrated on different aspects of biomass cofiring. Fernando (2005) reviewed the different types of biomass available and the experience of cofiring around the world. This was followed by a report concentrating on the cofiring of coal with waste fuels (Fernando, 2007). Fernando (2009) describes the current and emerging technologies for the co-gasification and indirect cofiring of coal and biomass. The interested reader is referred to these documents for more detailed information on these aspects of biomass use.

This report concentrates on the potential effect on non-CO₂ emissions from the co-combustion of biomass in coal-fired plants. This includes particulates, SO₂, NO_x, halogens, organic compounds and trace elements. The report also includes a section on potential changes in fly ash characteristics. Chapter 2 looks at the characteristics of different biomass types concentrating on the physical and chemical characteristics which could have an effect on emissions. Chapter 3 considers the behaviour of these materials in different combustion conditions and the

formation and/or release mechanisms for the different pollutant species. A few case studies on the effect of biomass addition on full-scale coal-fired plants are summarised in Chapter 4. Finally, Chapter 5 reviews the situation in several countries around the world that have some form of incentive, legislative or otherwise, to increase the practice of cofiring biomass with coal.

2 Biomass characteristics

Biomass is a term used to include any plant-derived organic matter available on a renewable basis such as dedicated energy crops and trees, agricultural food and feed crops, agricultural crop waste and residues, aquatic plants. However, it can also include animal wastes, municipal wastes, sewage sludge and other waste materials.

Most biomass materials are cellulose based and are therefore physically and chemically very different to coal. Biomass fuels typically contain a high percentage (up to 70–80%) volatile matter compared to the 10–50% present in most coals. Even though the ash content of biomass is commonly lower than that of coal, the content of alkali metals such as K and Na can be higher (Boneham, 2008).

The different fuels which can be used in biomass cofiring are reviewed in previous reports by the IEA CCC (Fernando, 2005, 2007). They consider the fuel characteristics and the cofiring experience in different regions with an emphasis on combustion operation and plant considerations such as slagging and fouling. The interested reader is referred to these earlier reports for further details. The following sections concentrate on those characteristics of biomass fuels which may affect the eventual emissions.

2.1 Wood

Wood, especially pelletised wood, tends to be low in ash and alkali content and therefore causes less deposition problems than other biomass materials. One Swedish plant (unnamed) has already converted from 100% coal combustion to 100%

wood combustion with relatively few operational challenges (Dai and others, 2008).

Wilk and Sarnowski (2009) studied the co-combustion of low ash (3–5 wt%), low sulphur (0.4–0.6%) Polish coal with wood pellets in a 25 kW laboratory combustor at the Silesian Institute of Technology in Poland. The wood pellets were 10 wt% moisture with low sulphur (<0.05 wt%), low nitrogen (0.04 wt%) and low chlorine (0.02 wt%). Figure 1 summarises the emissions and power output and how this varies depending on the share of wood fired in the system. Although the effects on different gaseous species are discussed in more detail in Chapter 3, it is clear that cofiring wood can have a positive effect in reducing emissions of all major pollutants from coal combustion without an overly detrimental effect on the power rating of the system.

Table 1 shows the average toxic metal levels of different woody biomass species, as studied by Demirbas (2005). It is difficult to compare these levels directly with the trace element concentrations of coals as coal trace element levels can be highly variable. But it is clear that the cofiring of woody biomass with coal is unlikely to be a problem with respect to increased emission levels. In fact, according to Demirbas (2005), emission levels listed for trace elements from most wood species are considered ‘very low’ and are often at or near the detection limits.

Sawdust tends to contain significantly lower concentrations of K, Mg and P than other biomass materials and this contributes to a higher concentration of metals in the ash (due to chemical interactions which are discussed in more detail in Chapter 3).

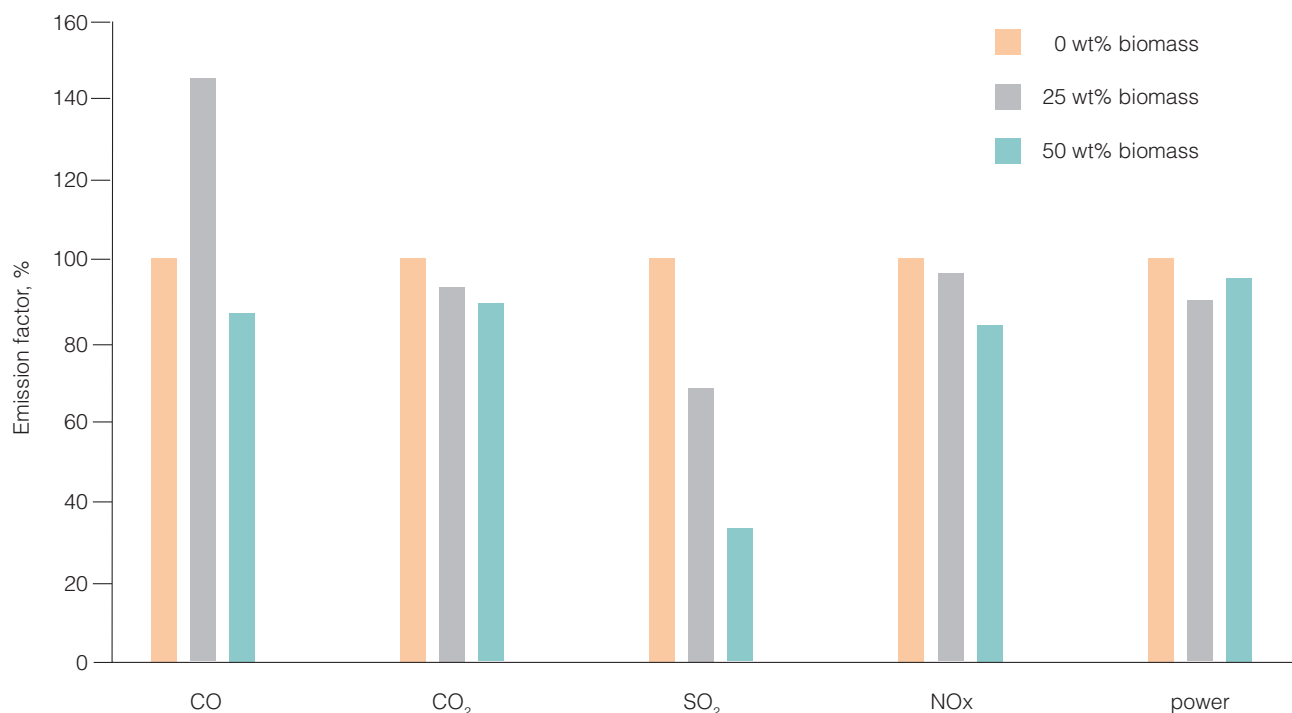


Figure 1 Dependence of the share of biomass (wood) on emissions and the power rating of a laboratory-scale combustion system firing Polish coal (Wilk and Sarnowski, 2009)

Table 1 Average toxic metal levels of different woody biomass species, mg/kg (Demirbas, 2005)

Biomass species	As	Cd	Cr	Cu	Pb	Hg
Spruce wood	0.033	0.013	0.022	0.413	0.324	<0.002
Beech wood	0.039	0.011	0.038	0.304	0.357	<0.002
Oak wood	0.024	0.020	0.036	0.382	0.268	<0.001
Spruce wood bark	0.034	0.014	0.030	0.448	0.376	<0.001
Beech wood bark	0.037	0.012	0.037	0.372	0.425	<0.001
Oak wood bark	0.019	0.016	0.035	0.415	0.312	<0.001

Sawdust from mainly bark, tree trunks or from young growth wood may contain significantly more (three times as much) Ca and K as other sawdust (Boneham, 2008).

Perhaps the main reason why wood is not cofired more often is the problem of how to introduce the wood into the combustion zone in a form that does not affect combustion efficiency. Compared to coal, wood has a low energy density, is heterogeneous and poses a challenge for the existing fuel handling systems. The previous IEA CCC report by Fernando (2005) discussed this in more detail. However, there are several ways of treating wood to make it easier to handle and this may, in turn, affect the combustion characteristics and the emissions. The Power Grade Charcoal project has been established between research organisations in the Netherlands, Sweden, Estonia, Germany and Poland. The project aims to promote the decentralised production of carbonised biomass for use in co-combustion. The carbonisation of wood to charcoal makes it much easier to handle and cofire than untreated wood. Charcoal was cofired in a 1:9 ratio with coal at the 600 MWe Electrabel power station in Planiec, Poland. The boiler emissions of particulates and NO_x were not affected by the addition of charcoal and the SO₂ emissions were actually reduced by 10%.

Torrefaction of wood is becoming increasingly popular as it overcomes many of the problems associated with wood as a fuel. During torrefaction, biomass is kept at 230–300°C over a short period of time (up to an hour) without oxygen. Although the resulting torrefied material is more stable and has a higher calorific value than the untreated wood, it still has a low bulk density. Pelletisation is therefore a common follow-up treatment (Romeo and Barno, 2008; Zanzi and others, 2008). Torrefaction of wood leads to enrichment of the trace element content due to the volatilisation of the material (van Eijk and others, 2009). However, since the trace element concentration of the wood is so low to start with, it is unlikely that any enrichment will lead to concentrations of trace elements that would be of concern.

2.2 Energy crops

Several plant species such as miscanthus, switchgrass and sorghum are now grown as energy crops for biomass combustion. Tabet and others (2008) compared the combustion characteristics of these crops with wood chips at

laboratory scale. These crops have a lower bulk density than wood and a lower moisture content at harvest. They tend to have higher ash content than wood – 5–7% dry matter for sorghum and switchgrass and 2–4% for miscanthus compared to 1–2% for wood. The nitrogen contents of the crops are comparable to that of wood. The sulphur content of the crops can be higher (0.1–0.2% for the crops compared to 0.05% for wood) but these values are still relatively low. The chlorine content of the miscanthus and sorghum (0.1–0.2%) is slightly higher for the crops compared to 0.05% for the wood. Switchgrass has the highest Cl content (at around 0.3–0.4% dry matter) which could lead to high HCl emissions.

According to Tabet and others (2008) the miscanthus and sorghum are suitable fuels for use in combustion but switchgrass may exceed boiler manufacturers recommended threshold values for Cl (normally set at 0.3%). With respect to their combustion characteristics, the laboratory study by Tabet and others (2008) suggested that miscanthus was the best of the three crops studied due to its ash fusibility, ash content and biomass yield rate.

The cofiring of switch grass blends at 15% (mass basis) at an unnamed 50 MWe plant resulted in a 20% reduction in NO_x emissions. It was also noted that there was ‘some traces of partially unburned switch grass in the ash’ (Dai and others, 2008) although it was not stated whether this had an effect on any ash sales.

Danish tests have shown that up to 20% (energy basis) straw can be cofired with coal without severe deposition or corrosion problems (Dai and others, 2008). Van Eijk and others (2009) note that, although grasses and straw are characterised by ‘high’ K and Cl content, these elements can be washed out of the biomass before combustion.

2.3 Waste agricultural materials

A literature review of cofired materials shows that a wide range of waste agricultural matter has been tested in co-combustion. The characteristics of these materials can be quite unique. For example, **olive pellets** may contain significantly more (2–3 times as much) K and Ca than other biomass materials (Boneham, 2008). **Date stones** are of considerable importance as a biomass fuel in regions such as North Africa and Middle Eastern countries. The date stones

tend to be 7% moisture, 69% volatile material, 23% fixed carbon and only 1% ash. Combustion of date stones in a small furnace was compared with coal combustion under the same conditions. The date stones showed higher combustion and heat transfer rates than the coal, due to the higher volatile matter content and lower ash content, although the heating value of both fuels was around the same (around 25 MJ/kg) (Al-Omari, 2009).

Palm kernels are reported to have ‘significant’ nitrogen content at 2.9% (dry, ash free value, Boneham, 2008) while **meat and bone meal** can be ‘high’ in Cl (Dai and others, 2008). **Chicken litter** is heterogeneous in nature and can therefore result in more erratic combustion behaviour than some other biomass materials. The ash in chicken litter can also be high in alkali and alkaline earth metals (Di Nola and others, 2009).

Dairy biomass (collected manure and soil/dust from cattle sheds) has a lower heat value than coal due to less fixed carbon, more oxygen, more fuel bound nitrogen and more ash. On a dry, ash free basis, dairy biomass has around 60% of the heat value of coal. The high ash value of the dairy biomass means that blends are limited to less than 10% on a heat basis (Lawrence and others, 2009).

2.4 Sewage sludge

Sewage sludge is an important biomass material but is also one of the most challenging with respect to potential emissions. In 2005, around 9 Mt of sewage sludge was produced in 2005 in Europe and 7.5 Mt in the USA. Germany has possibly the greatest experience with 17 plants having performed trials cofiring sewage sludge and 10 plants continuing to do so. Changes in EU legislation have meant that sewage sludge can no longer be sent to landfill (see Chapter 5). There is therefore a move towards optimising sewage sludge for cofiring in coal combustion systems. The high content of organic matter makes it ideal as an energy source but the high moisture content poses a technical challenge. Pre-treatment such as drying and conversion to a WDF (waste derived fuel) is necessary (Barbosa and others, 2009).

Sewage sludge can have significantly higher concentrations of sulphur than coal due to sulphur-containing compounds used in the wastewater treatment plant for flocculation of the sludge. The iron content of sludge can also be high (Amand and others, 2001).

Sewage sludge contains large quantities of nitrogen which can result in high concentrations of NO which often must be abated with flue gas treatment. However, the cofiring of coal with sewage sludge in fluidised bed combustion (FBC) systems has been shown to effectively reduce the NO emissions. This is due to the char introduced by the coal actively reducing the NO (Leckner, 2007). FBC systems are discussed in more detail in Section 3.

Table 2 shows the difference between emissions from firing coal and from cofiring sewage sludge at the 760 MWe, Unit 7

of EnBW’s Heilbronn plant in Germany. The sewage sludge is dewatered, dried and fired at 4%, by weight, <1.1% by thermal input. The plant has been in continuous operation since 1998, and since then over 15,000 t/y sludge has been cofired. As shown in Table 2, the emissions of CO, SO₂ and NO_x are generally similar for coal and coal and sewage sludge cofiring. The emissions of particulates and halogens have dropped as a result of cofiring whereas emissions of organic carbon may be raised at certain times. Emissions of trace elements and heavy metals such as Cd and Hg do not appear to be affected significantly (Buck and Triebel, 2000).

Cofiring sewage sludge (9% by weight, <0.5% by thermal input) and waste wood (unspecified proportion) at the Stadtwerke 252 MWth CFBC boiler in Duisburg, Germany, was shown to result in reduced emissions and actually improved the operation of the plant. It was also possible to reduce the need for additional limestone when cofiring sewage sludge (Fernando, 2007).

The ashes produced from the cofiring of sewage sludge with coal have been shown to contain higher contents of heavy metals than those from coal combustion alone. A literature review by Barbosa and others (2009) indicated that several studies showed an increase in emissions of Cd and Hg. There is also concern that the leachability of trace elements is greater in ash from systems cofiring sewage sludge. This is discussed in more detail in Section 3.3.

2.5 Other waste materials

The main concerns with cofiring waste materials appear to be whether cofiring these materials may cause damage to the plant (for example through corrosion or slagging and fouling) and also whether the use of these materials means that the coal plant now has to meet new emission standards or emission standards that merge standards for coal-fired plants with those of waste incinerators. This is discussed in more detail in Chapter 5.

Characterising refuse derived fuel (RDF) is not simple since the material itself is so variable. RDF is produced by processing waste through shredding, screening, magnetic and current separation followed by drying and compaction into a pellet cube. The materials used in RDF can be optimised to promote combustibility. For example, rubber and non-chlorinated plastic can be added to increase the calorific value of the fuel. It has been suggested that RDF may have elevated phosphate levels which could affect fly ash quality but this has not been reported in practice. RDF can often contain high concentrations of chlorine (0.8%) which can cause plant corrosion issues but does not seem to result in high chlorine emission problems (Fernando, 2007).

The lime concentration in **paper mill sludge** can lead to a reduction in emissions of SO₂ from cofiring coal in a circulating fluidised bed (CFBC) system. The moisture in the sludge also results in a lower combustion temperature which reduces the NO emissions (Leckner, 2007).

Around 1 billion **waste tyres** are generated worldwide every

year, equivalent to about 2% of total annual solid waste. Tyres are resistant to chemical, biological and physical treatments for disposal and are therefore a significant waste issue. In regions such as the EU, the disposal of tyres in landfill is no longer an option. The cofiring of tyres with coal has been shown to be an efficient way to generate power from waste and also reduce emissions of NO_x. Shredded waste tyres are an excellent reburn fuel and have low nitrogen and chlorine contents along with a high calorific value. In some countries, such as the UK, the definition of tyres as waste material may mean that this fuel cannot be used in coal-fired power plants as a cofired fuel. However, the indications are that this definition is inappropriate and that a change of classification would allow tyres to be a useful source of energy in future (Singh and others, 2009). In the USA, some air regulations include plants firing tyres as solely fossil-fuel fired plants. Tyres usually contain a significant amount of substances made from fossil fuels (carbon black and/or synthetic rubber) and so there is some debate over their classification in some areas as 'renewable'.

2.6 Comments

The characteristics of biomass and waste materials are variable with each posing a different challenge with respect to suitability as a cofiring fuel. However, these challenges are largely associated with the logistics of the delivery of the material to the plant and into the combustion system, and with potential slagging, fouling and corrosion issues at the plant. There seem to be few, if any, issues being raised with respect to increased emission levels of major elements such as particulates, SO₂ or NO_x. In fact, the increased efficiency of combustion and the presence of emission control systems usually means that detrimental emissions from waste combustion alone can be reduced or avoided and this will be discussed in more detail in Chapter 3.

In some cases, the inclusion of waste fuels such as RDF can mean that a coal-fired plant must now meet new standards which reflect the inclusion of waste materials. This is discussed in more detail in Chapter 5.

3 Effects of cofiring on emissions

Cofiring of biomass with coal is commonly achieved by retrofitting existing coal-fired plants with plant-specific modifications to allow efficient feeding of the biomass fuel into the boiler. These modifications typically include alterations to the fuel-processing, storage and delivery systems. Cofiring can be achieved in most types of coal-fired boilers including stokers, fluidised beds, pulverised coal fired boilers and cyclones (Eisenstat and others, 2009). Kangas and others (2009) agree that cofiring of biomass with fossil fuels has been demonstrated in essentially every boiler type and is commercially used in many areas, including in multi-fuel boilers in Scandinavia. The amount of biomass used in cofiring relative to the total use of biomass for power production is considered to be relatively small, although there are no global statistics available.

Current best available technologies for coal have conversion efficiencies of up to 46–47% – that is, up to 46–47% of the theoretical energy contained in the coal is actually converted to produced electricity (based on the net calorific value of the coal). New technologies hope to increase this to more than 50%. In a 10 MWe dedicated biomass plant, the efficiency could be up to around 35% (Olivera, 2008). The cofiring of biomass with coal offers the potential for more efficient combustion of the biomass material through the higher combustion temperatures and optimised combustion conditions associated with the combustion of coal. However, the addition of biomass to a previously dedicated coal combustion unit has the potential to alter combustion conditions in a detrimental manner. The difference in burn efficiency and combustion chemistry could have an effect on particulates and other emissions.

According to Koppejan and Baxter (2005) over 150 coal-fired plants worldwide have experience of cofiring with biomass or

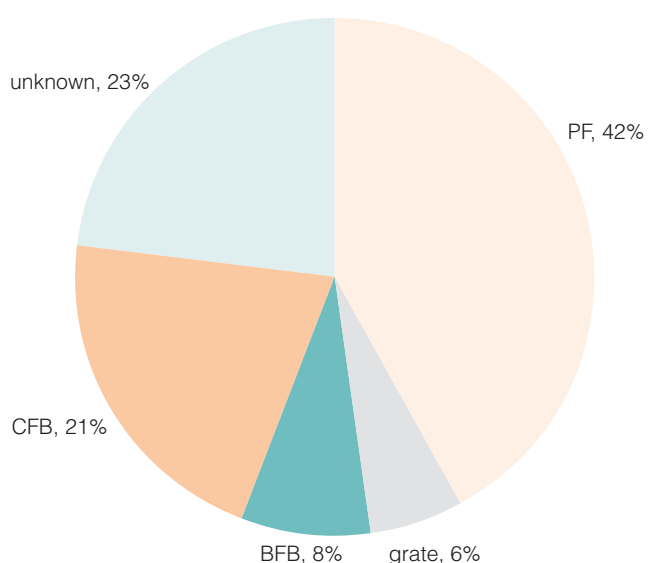


Figure 2 Distribution of firing systems with coal-fired power plants that have experience with cofiring biomass (Koppejan and Baxter, 2005)

waste, at least on a trial basis. The greatest number of tests have been in the USA (41), Germany (27) and Finland (18). Figure 2 shows the breakdown of tests by plant type. The majority of tests have taken place on pulverised coal fired plants, probably because these are the most common type of large-scale combustion systems for coal. However, despite the relatively high number of tests, only 40 or so plants currently cofire biomass on a commercial scale. This report concentrates on emissions from full-scale pulverised coal combustion systems, although some data for the grate-fired, fluidised bed and gasification systems are included for comparison.

As mentioned in Chapter 2, the characteristics of each type of biomass can be quite distinct and each can pose its own challenges or provide unique benefits. It can be seen from a review of the literature that most trials of new biomass materials as co-combustion materials are considered unique and worthy of publication. In order to provide a valid representation of this variability, it is necessary to review a range of co-combustion trials.

3.1 Pulverised coal combustion systems

Biomass can contribute from 2% to 30% of the total heat needed for generation in an average coal-fired boiler. The amount of material cofired will commonly be determined by factors such as the availability of the fuel, the ease of handling, air emission limits and any potential detrimental effects on plant performance and plant output.

When comparing the combustion characteristics of coal versus biomass in full-scale pulverised coal plants, Demirbas (2004) makes the following conclusions:

- pyrolysis starts earlier for biomass compared to coal;
- the volatile matter of biomass is higher than that of coal;
- the fractional heat contribution from the volatile matter is 70% for biomass compared to 36% for coal;
- biomass char has more oxygen compared to coal;
- the heating value of volatiles is lower for biomass fuels;
- pyrolysis of biomass chars mostly releases CO, CO₂ and H₂O;
- biomass fuels have ash which is more alkaline in nature.

The majority of these characteristics can be considered to be more important with respect to plant performance than they are to emissions. The most negative effects due to the different chemistry of biomass fuels tend to be slagging and fouling of the power plant itself. This is dealt with in more detail in a separate report from the IEA CCC (Barnes, 2009).

In a previous IEA CCC report, Fernando (2005) summarised the effects of biomass cofiring on combustion systems including the potential for the deactivation of selective catalytic reduction (SCR) catalysts due to increased concentrations of alkali metals such as As, P or Fl. The lower

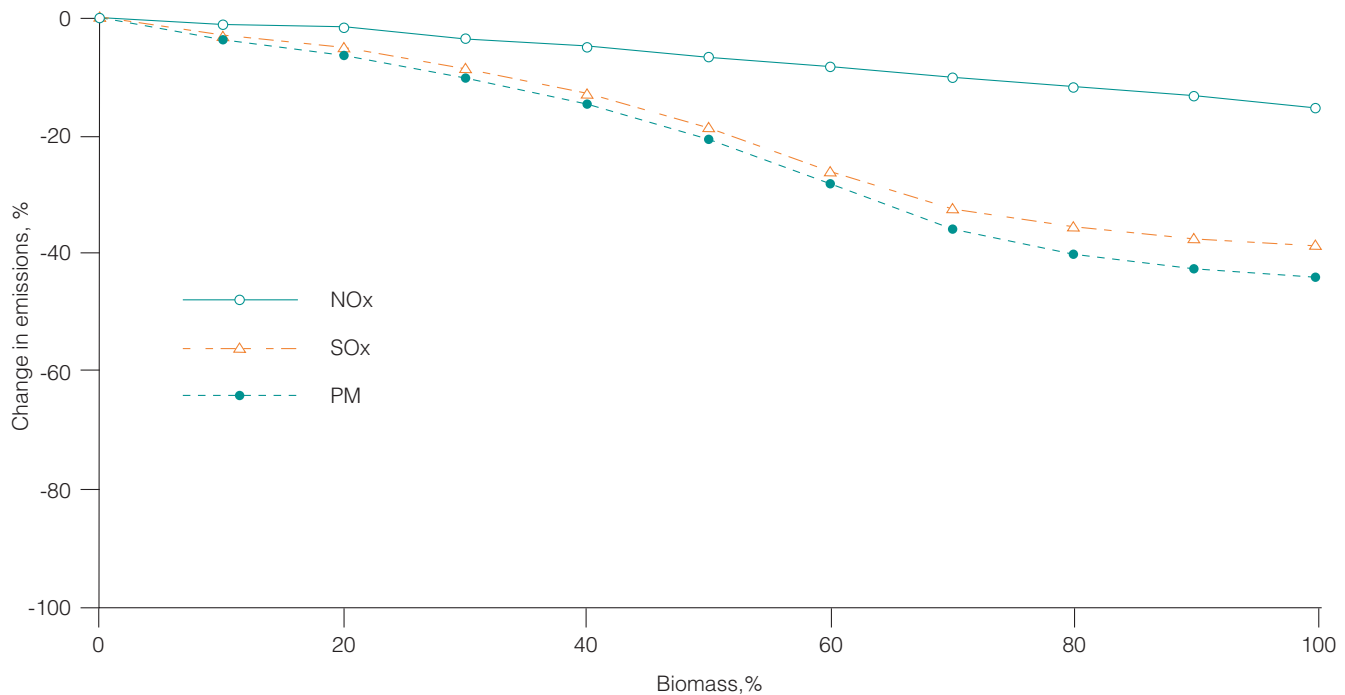


Figure 3 Plots for average emission impacts of cofiring coal with biomass (Demirbas, 2005)

sulphur content of most biomass fuels will lower the load on the flue gas desulphurisation (FGD) plant but there is also the possibility that any increase in HCl emissions will impair the operation of the FGD systems. Further, any increase in heavy metals such as Hg, As or Pb will end up in the FGD residues. These effects are beyond the scope of this report. The remainder of this Section concentrates on the effect of biomass combustion on flue gas emissions.

Pulverised coal combustion systems are characterised by high combustion temperatures (>1200°C) and relatively high combustion efficiency. This is certainly the case when comparing pulverised coal fired boilers to the average dedicated biomass combustion system. This means that, in general, the conditions provided by the pulverised coal fired boiler promote more efficient and complete combustion of biomass. The cofiring of biomass with coal enhances the performance of the biomass combustion but will normally result in at least a small reduction in plant output compared to coal firing alone. However, it would appear from the literature that these effects are generally minimal. For example, cofiring sawdust with coal (up to 12% on a heat basis) in the 32 MWe Seward Station in Pennsylvania, USA, had a 'slight' effect on the unburnt carbon along with a 0.5% decrease in boiler efficiency. However, the effect on CO₂, SO₂ and NO_x emissions was reported to be 'favourable' (Dai and others, 2008).

Actual changes in emissions due to cofiring biomass in pulverised coal systems are covered in the separate sections below. However, it appears to be fair to say that, in the majority of cases, the addition of biomass materials to pulverised coal fired plants has a beneficial effect. For example, Dai and others (2008) cite a somewhat dated but still relevant reference from 1999 where the cofiring of high moisture (19 wt%) biomass resulted in a reduction of SO₂ and NO_x emissions (by 17% and 2–3% respectively). The majority of papers published since 1999 would seem to agree

that emissions of SO₂ and NO_x are reduced with the addition of biomass.

Demirbas (2004, 2005) gives excellent reviews of the effects of cofiring biomass with coal on emissions and Figure 3 summarises the average impact on emissions. Although this figure can be regarded as a generalisation, the trends summarised therein do seem to be confirmed by the majority of the studies reviewed in this report. That is, assuming that combustion conditions are optimised, the cofiring of biomass with coal results in a reduction of emissions of major pollutants such as SO₂ and NO_x. In fact, Dai and others (2008) cite over 12 papers which all agree that blending coal and biomass can lead to reductions in pollutant emissions.

Figure 4 summarises the synergistic effects that the properties of biomass fuels can have on emissions from cofired systems (Leckner, 2007). It is clear from the crossed lines that many of the elements present in the fuels can have an effect on the formation of several emitted species. The relative concentration of each of these elements and their interactions with other species during the combustion process will determine whether the effects on emissions are positive or negative. Either way, since the elements must end up in one waste stream or another, significant changes in the combustion chemistry will affect either the flue gas emissions, the ash from the particulate control systems, or the elements will remain within the plant and cause deposition and slagging problems.

In the USA, CO (carbon monoxide) is included as a criteria pollutant and relevant emission limits apply. This is not the case in the EU where CO emissions are often monitored to ensure efficient combustion conditions but are not subject to defined emission limits. Emissions of CO tend to be associated with combustion systems which are not running correctly as it is an indication of incomplete or inefficient combustion. The co-combustion of biomass in coal systems can lead to increased CO emissions as a result of changes in

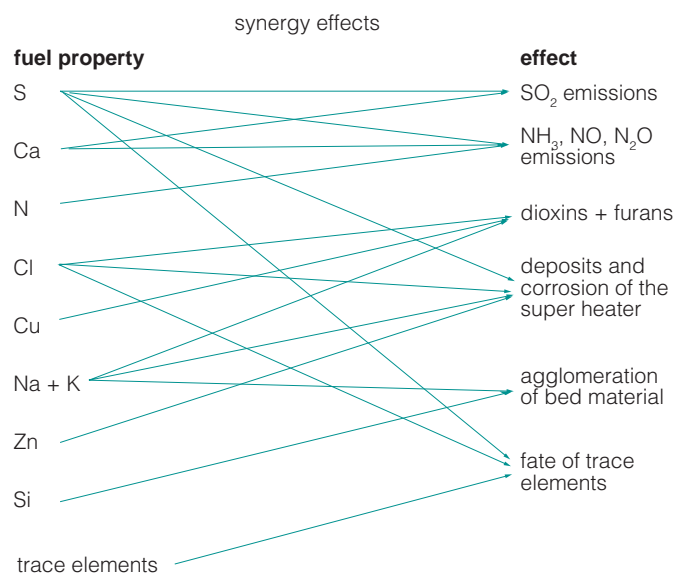


Figure 4 Synergy effects between cofired fuels (Leckner, 2007)

combustion conditions and this will be an important consideration for plants in the USA which are considering co-combustion.

The following sections summarise the generalisations which can be made with respect to cofiring biomass with coal. Most biomass materials tend to result in the same effects on emissions (for example, most biomass results in lower SO₂ emissions). However, as discussed in Chapter 2, biomass and waste species vary considerably in their chemistry and so there are exceptions to the rule. These will be discussed.

3.1.1 SO₂

According to previous IEA CCC review (Fernando, 2005), SO₂ emissions ‘invariably’ decrease during co-combustion, often in proportion to the amount of biomass used. Biomass generally has a lower sulphur content than coal and will therefore contribute to lower SO₂ emissions. Waste materials (organic) also tend to have low sulphur contents (Leckner, 2007) as does municipal solid waste (MSW) (Fernando, 2007). Figure 5 shows the consistent reduction in SO₂ emissions with different blends of different fuels (Spliethoff and others, 2000). The only exception is sewage sludge. The sulphur levels in sewage sludge range from levels which are similar to those for coal to levels which are significantly greater. Sulphur concentrations in sewage sludge can be elevated as a result of the flocculation treatments used during processing. Whether this results in increased SO₂ emissions depends largely on the levels of alkali and alkaline earth compounds present in the ash which can help trap SO₂. Although sewage sludge contains high levels of CaO this does not guarantee significant sulphur capture in the ash during sewage sludge combustion. It has been suggested that, during the co-combustion of sewage sludge with coal, the higher combustion temperatures may inactivate the surface of the CaO which inactivates its sulphur-capturing capabilities (Fernando, 2007).

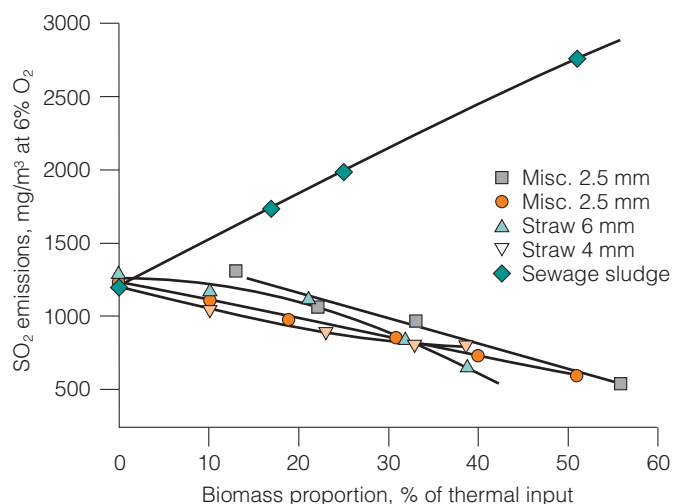


Figure 5 Emissions of SO₂ as a function of biomass ratio for several blends (Spliethoff and others, 2000)

The sulphur content of tyres (0.9% on a dry basis) is higher than that of some coals but not all. This could mean a slight increase in sulphur emissions from co-combustion. However, the increase is likely to be minimal (<10%) and could be handled by existing FGD units.

3.1.2 NO_x

As discussed in previous IEA CCC reports (Sloss, 1998a; Wu, 2002), NO_x emissions arise through two different pathways during combustion:

- fuel emissions – N in the fuel gets released;
- thermal emissions – N₂ from the combustion air becomes oxidised.

It is not possible to easily predict NO_x emissions, even when the fuel N content is known as some of the fuel N may not ultimately be released as NO_x but rather as other volatile compounds. The combustion conditions play a far greater role in determining NO_x emissions than the N content of the fuel. However, the characteristics of the fuel will determine the combustion conditions and ultimately affect NO_x emissions.

Leckner (2007) suggests that some large-scale co-combustion tests concern such low quantities of biomass (<10%) that the impact of co-combustion on NO can hardly be noticed. However, it would seem from the available literature that most studies note a significant reduction in NO with biomass co-combustion.

The nitrogen content of biomass is generally lower than that of coal leading to lower fuel-NO_x emissions. RDF may also have lower N contents than average coals whereas sewage sludge can contain higher N than most coals. However, despite the lower fuel nitrogen in most biomass materials, NO_x emissions per unit energy input from biomass combustion alone are reported to be higher than those from coal combustion alone. This is due to the formation of greater amounts of thermal NO_x emissions during the less efficient combustion of biomass (Dong and others, 2009).

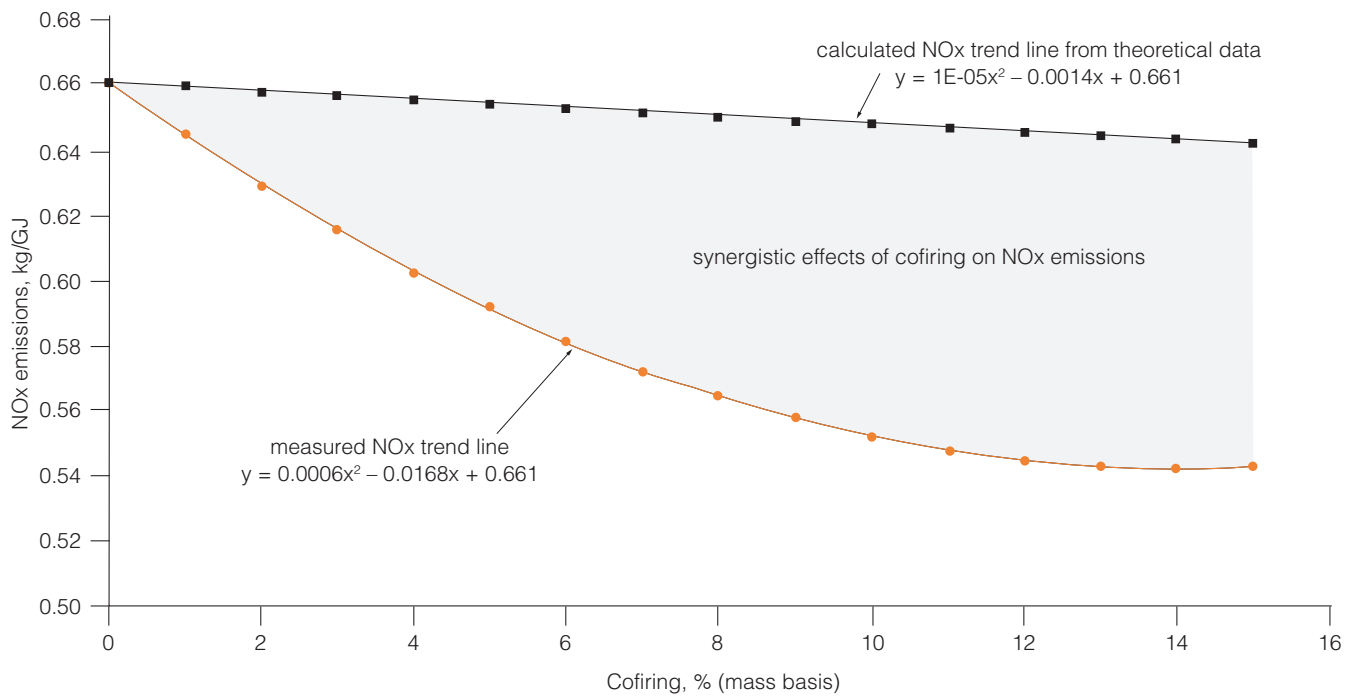


Figure 6 Synergistic NOx reduction from cofiring biomass (Lawrence and others, 2009)

The NOx emissions from cofiring waste with coal are a combination of fuel and thermal NOx emissions and can vary considerably depending on the fuels and combustion conditions. Di Nola and others (2009) note that, with different fuels, the volatile N species emitted do not correlate with the initial fuel nitrogen content. They also noted that biomass pyrolysis resulted in higher volatile N yields than coal. The mechanism of conversion of fuel N into NOx species is still not fully understood. However, it is suggested that the partitioning of the fuel nitrogen during devolatilisation is important in influencing the NOx species formed. In combustion systems, emissions are reported to be lower if the fuel N is released with the volatiles rather than retained in the char. This is because volatile nitrogen can be controlled with low NOx technologies such as air partitioning and fuel reburning. Since biomass pyrolysis results in higher volatile N yields, this would imply that NOx control in biomass systems is easier than that in coal-fired systems and that biomass cofiring has an 'intrinsic NOx reduction potential'.

Over and above this volatility effect, biomass cofiring can lower flame temperatures reducing thermal NOx emissions. Further, the presence of NH₃ in some biomass materials, such as animal wastes, or formed during the combustion of biomass can contribute to the catalytic reduction of NOx (Dai and others, 2008).

Lawrence and others (2009) summarised previously published data into the results shown in Figure 6. The results are from cofiring coal with low N agricultural waste. The top line of the graph shows the predicted reduction in NOx emissions with increased proportion of waste material, based on the lower N content of the waste material. However, the actual measured NOx emissions were much lower. This is assumed to be due, at least in part, to the increased volatile matter in the waste material (46.88% in the dairy biomass compared to 28.49% in the coal) causing the rapid reduction of O₂, which in turn reduces the rate of formation of NOx from fuel nitrogen.

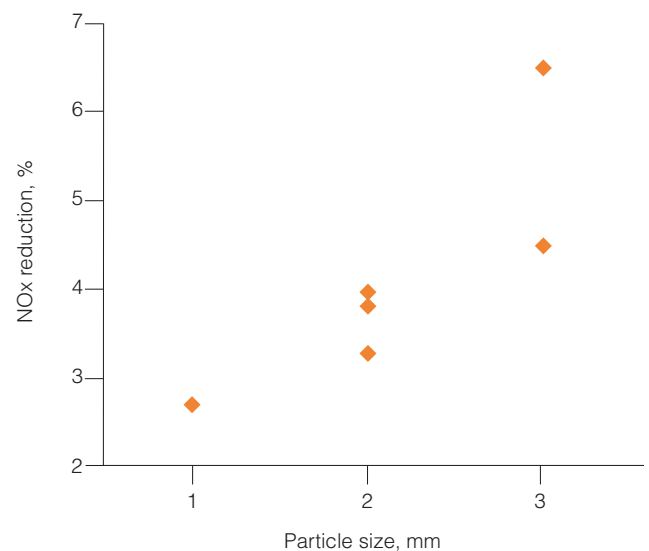


Figure 7 NOx reduction against biomass particle size fed into the boiler (Canalis and others, 2008)

The emissions of NOx from combustion systems have also been shown to be affected by the particle sizes of the biomass material in the cofiring system. Figure 7 shows the increase in NOx reduction with the increase in particle size. The figure demonstrates that the larger the particle size, the higher the NOx reduction rate. Figure 8 shows the NOx reduction variation with increasing particle size against the percentage reduction in O₂ (excess air reduction). The greater the reduction in excess air, the greater the NOx reduction. These data were obtained from biomass combustion at a full-scale plant but no more detailed information was available (Canalis and others, 2008).

The NOx reduction potential of cofiring biomass with coal is so effective that the co-combustion of waste with coal can

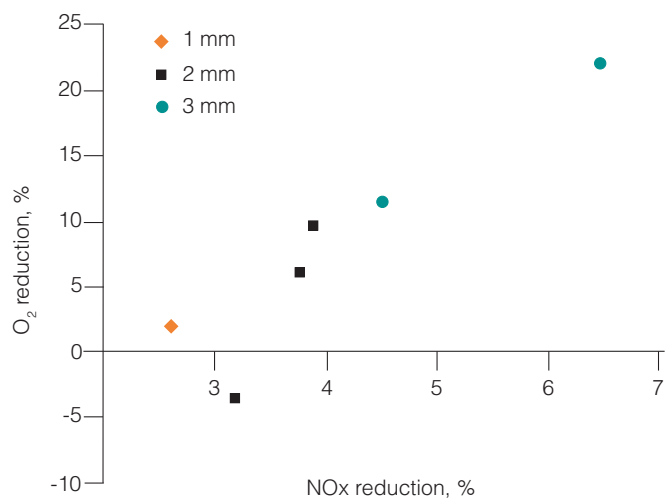


Figure 8 NOx reduction against excess air reduction (Canalis and others, 2008)

actually be used as a NO_x reduction strategy. For example, the high calorific value and the low nitrogen content of waste tyres make them ideal as a reburn/cofiring fuel to reduce emissions of NO_x from coal combustion. Tests on an 80 kWth plant in the UK, cofiring South African coals with shredded waste tyres (thermal input of over 12%) showed that the tyres were more efficient as a reburn fuel than coal (35% more efficient) or gas (53% more efficient at NO_x reduction). This is due to the higher volatile hydrocarbon content of the tyres compared to coal (Singh and others, 2009).

Emissions of NO_x from biomass combustion alone or from biomass cofiring can be reduced by 10–20% using fuel or air-staging technologies. Dong and others (2009) describe the development of a decoupling combustion technology which separates combustion into two stages – the pyrolysis of the fuel and then the combustion of the fuel and the pyrolysis gas. Coal blends from the Datong and Shanxi provinces of China were fired with sawdust, rice husk and corn straw from the Heilongjiang province in a 10 kW stove manufactured with the decoupling technology. Tests were run in both a traditional combustion system and the decoupled system at various biomass blend rates – 0, 15, 50, 75 and 100% on a mass basis. In the traditional combustion system, the maximum NO emissions were seen at a 25% mix for sawdust and rice husk but 50% for corn straw. Although the trends in NO emissions were generally the same for the biomass mixes in the decoupled combustion system, the emissions were significantly lower with an average of 19% reduction for all fuels.

Deep air staging conditions can also be used to reduce NO_x emissions from co-combustion systems. Deep air staging is a combination of advanced low NO_x burners with multiple overfire air ports in the furnace. Cieplik and others (2008) have shown, at laboratory scale, that emissions of NO_x from coal combustion can be reduced by up to 40% with deep air stage conditions. Emissions of NO_x from cofiring biomass with coal are already lower than those when coal is fired alone, but deep air staging can reduce emissions further. This is due to the higher volatility of the biomass fuels producing more reducing conditions in the flame, without the application of additional staging. Overall, the NO_x reduction in the cofiring test with deep air staging was more than 50%.

Although all the evidence reviewed for this report suggests that cofiring biomass with coal will virtually guarantee reductions in NO_x emissions, there is one potential problem. It has been reported that waste cofiring can affect NO_x emissions by adversely affecting the performance of SCR catalysts. Many biomass/waste materials contain alkali or alkaline-earth metals (such as potassium chlorides and sulphates) which interact with SCR catalyst activity. A previous IEA CCC report by Fernando (2007) deals with this in more detail but suggests that the location of the SCR is important, with those in a high-dust configuration being more likely to be affected. However, there does not seem to have been any published material specifying how best to either avoid or reduce this potential effect.

3.1.3 Halogens

A previous report from IEA CCC (Davidson, 2005) reviews the effect of cofiring on chlorine emissions, concentrating mostly on the chemistry in the combustion zone and the effect on corrosion within the plant. One of the main conclusions was that cofiring coal with biomass could reduce the potential for corrosion that would be encountered with firing biomass alone. However, the addition of biomass to coal combustion systems could result in increased halogen concentrations and this may lead to problems with corrosion, and slagging and fouling that would be considered severely detrimental for a full-scale coal-fired plant. This was dealt with in detail in the previous IEA CCC reports by Fernando (2007) and Barnes (2009).

The concentration of chlorine in coal can be somewhat variable but the concentration of chlorine in biofuels can be much higher or much lower than any concentrations found in coal (Leckner, 2007). The Cl contents of RDF and MSW can be up to 1% Cl which is ten times greater than in typical bituminous coals. Sewage sludge has a lower Cl content.

Some biomass fuels, especially straws and grasses, can have a higher percentage of Cl than coal (Eisenstat and others, 2009). Although this can lead to problems within the plant, there have been no reports of increased emissions of these species from cofiring. Montgomery and others (2008) report on ‘considerable’ high temperature corrosion problems caused by the high content of potassium chloride in biomass cofired with coal. The potassium chloride is converted into potassium sulphate in the combustion chamber and sulphate rich materials are deposited on metallic surfaces such as high temperature super heaters. The Studstrup power plant unit 4 in Germany was run initially for 3000 hours using 0–20% straw with coal and reported problems with corrosion. This was reduced when the fuel mix was lowered to 10% straw with coal.

The sulphur and aluminosilicates present in coal can help to prevent Cl deposition within the plant. If the chlorine content remains an issue, as may be the case with some materials such as straw, the biomass can be pre-treated with water which can have a concomitantly beneficial effect on ash fusion temperatures (Dai and others, 2008).

It does not seem that emissions of chlorine from biomass

cofiring cause any major problems as there is little or no indication of this in the published literature. No data were found reviewing chlorine emissions from co-combustion which did not concentrate wholly on slagging and fouling issues. It is therefore assumed that emissions of chlorine and other halogens from the flue gases of co-combustion systems are not significant enough to warrant treatment. However, the presence of elevated chlorine in the combustion and flue gases can have a significant effect on the behaviour of trace elements such as As and Hg, as discussed in Section 3.1.4. In fact, the effect of the higher concentration of chlorine in biomass on the higher concentration of mercury in the coal is an attractive effect when considering which fuels to cofire. Low rank coals such as the Powder River Basin subbituminous coals are generally low in chlorine. This means that the mercury present during combustion is in the elemental state and is not captured efficiently in any existing pollution control systems. Mercury emissions from these low chlorine, subbituminous coals pose more of a challenge for mercury reduction than other coals. There are several plants in the USA which now blend coals specifically based on the chlorine content to ensure maximum Hg reduction (Sloss, 2002, 2009).

3.1.4 Trace elements

A previous IEA CCC report by Sloss and Smith (2000) reviews the behaviour of trace elements in coal combustion systems. The partitioning of trace elements during combustion is dependent on many factors such as:

- the size distribution of the coal/fuel;
- the combustion conditions;

- the modes of occurrence of the trace elements in the fuel;
- the interactions with other species in and after the combustion zone.

Therefore, although the trace element concentration of the fuel/biomass is important with respect to the emissions, it is not necessarily a predictor of increased emissions. Van Eijk and others (2009) found that, in both pilot- and full-scale studies, the speciation of trace elements depended on the type and composition of the biomass but also on the redox potential of the flue gas.

The volatilisation of trace elements from biomass is different to that from coal. However, those elements which are prone to volatilisation, such as S, Hg, Pb and Zn, will volatilise in both cases and, once volatilised, their behaviour in the combustion zone will be the same. Some trace elements, such as Mn and Cr, show differences in volatilisation between the different fuels and therefore this may affect the final emissions.

Trace element concentrations of different fuels are summarised in Table 3 (Mukherjee and others, 2003). The co-combustion of 10% car tyre scrap with the usual coal and/or petroleum coke mix at a cement plant may reduce emissions of Hg and Cd+Tl due to the lower concentrations of these elements in tyre material. However, emissions of Cr, Mn, Fe and Zn may increase (Sloss, 2007).

As shown in Table 3, the concentrations of some trace elements can be significantly higher in some biomass materials than in coals. To date, there has been nothing published which suggests that these elevated concentrations

Table 3 Selected trace elements concentration in different fuels, mg/kg (Mukherjee and others, 2003)

Element	Coal	Oil	Petcoke	Paper sludge	Municipal sewage sludge	Waste wood	Biomass mix*	Car tyre
As	2.6	0.02	1.1	3.2	8.4	10.0	2.3	0.65
Cd	0.10	0.2	0.2	0.53	3.79	1.4	0.70	<2
Cr	17	3.0	18	18	113	78	26	0.025
Cu	10	2.5	1.8	98	406	135	57	–
Hg	0.11	0.01	0.02	0.24	3.28	0.17	0.06	<0.1
Mn	41	2.5	5.7	6.3	546	92	157	–
Ni	12	120	278.6	10	83	31	16	0.013
Pb	6.7	2.0	2.1	31	260	574	16	0.005
Sb	0.51	0.02	0.6	1.2	4.1	16	1.6	–
Sn	1.4	0.002	0.6	6.2	38	6.4	1.3	–
V	24	180	1560	5	24	10	6	–
Zn	19	4.0	7.0	464	1349	807	133	15300
Calorific value, MJ/kg	24.4	32.0	35.0	10.98	13.14	11.86	–	36.0

* biomass mix = green wood, garden waste, straw, roadside grass and manure

have led to compliance issues in existing plants that have switched to cofiring (for example, those units discussed in Chapter 4). However, emission standards in the EU, USA and elsewhere are tightening and it could be the case that certain biomass materials could pose a problem with respect to maintaining emissions of some trace elements below legislated limits in the future. The trace element concentrations of biomass materials can be highly variable and there is also the possibility of significant variation between the characteristics of different batches of biomass from different sources, since biomass materials often reflect differences in trace elements in local soils and waters in the area of production. It is therefore likely that many plants switching to cofiring biomass materials will be required, under the operating permit or relevant requirements for the plant (depending on the location of the plant and the applicable legislation), to demonstrate that the new fuel material will not cause an increase in emissions of trace elements. In some cases, this may result in an increased amount of emissions monitoring. Monitoring will be required following the introduction of the new fuel to ensure that the emissions are not elevated to a level of concern. Over and above this, it may be necessary to perform some form of evaluation of the variability of the trace element concentrations of the biomass materials to guarantee that the emissions will remain below any legislated limit. If there is doubt, it could be the case that regular or continuous monitoring for any species of concern could be required. The plant manager would then need to make a decision on whether the added monitoring requirements and associated costs would override any benefit from the introduction of co-combustion at the plant.

Contreras and others (2009) have shown that predicting As emissions from cofiring coal with biomass materials requires relatively complex modeling. As mentioned in Section 3.1.3, the chlorine content of the fuels can be significant – the chlorine can increase As vaporisation and therefore increase the gaseous emissions of this element. Conversely, higher sulphur in the coal will increase the retention of the As in the ash. Calcium in the fuel mix will also increase As retention in the ash. Trace elements can interact with each other to affect whether they end up in the ash or are emitted in the flue gas. For example, Cd can interact with As to increase As retention in ash. Silicon may increase As volatility and reduce the efficiency of the interaction of As with calcium in the ash.

Chao and others (2008) note that the selenium content of coal (9.4 mg/kg for the sample studied) was consistently higher than the selenium content of rice husk (4.43 mg/kg) and bamboo (7.50 mg/kg). The arsenic concentration of coal was also higher – 7.58 mg/kg for coal compared with 2.30 mg/kg for rice husk and 0.384 mg/kg for bamboo. Mercury was not detected in either of the biomass samples.

The trace element of most concern with respect to emissions from coal and biomass combustion at the moment is mercury. A full-scale demonstration rig at the EU TOMORED project in Germany showed the increase in mercury input to the boiler with the cofiring of 10% sewage sludge. The Hg input to the plant increased from 0.12 mg/kg with just coal to 0.13 mg/kg with the sewage sludge/coal mix. There was no

increase in mercury in the bottom or ESP ash or gypsum but an increase in mercury in the waste water (from 0.001 to 0.002 mg/L) and an increase in emissions from the stack (2.2 $\mu\text{g}/\text{m}^3$ to 3.1 $\mu\text{g}/\text{m}^3$) (Thorwarth, 2006). Conversely the co-combustion of straw at the same plant showed increased Hg oxidation and removal across the ESP to the extent that the authors suggested that the cofiring of straw with coal could actually reduce mercury emissions to the air (Thorwarth and Scheffnecht, 2006).

The trace element contents of a range of biomass materials tested and/or used at Drax Power Station in the UK are listed in Table 4. The limits set by the UK Environment Agency specifically for the Drax plant are also shown in the table. In some cases, the trace element content of some fuels was near the set limit. For example, the plant specific limit of 0.1 mg/kg for mercury could have been reached or even exceeded by the milled palm nut. The Pb content of the milled palm nut was also approaching the prescribed plant limit. However, measurement of emissions around the plant and in local areas suggested that the emissions were below any level that would cause environmental effects and that perhaps the limits that had been prescribed could be ‘eased without any environmental detriment’. The easing of the limit could also mean that other biomass materials could become acceptable for use.

The Avedøre 2 main coal-fired boiler in Denmark cofired wood pellets with heavy fuel oil and gas. Iron vanadates were noticed in the reaction products as a direct result of vanadium introduced in the fuel oil. High Va is typical of oil but not of coal (Montgomery and others, 2008).

3.1.5 PAH

Polycyclic aromatic hydrocarbons (PAH) can be formed during combustion of fuels by either pyrolysis (the destruction of larger organic compounds) or pyrosynthesis (by the recombination of smaller molecules).

It is understood that biomass combustion, especially in less efficient systems such as domestic stoves, can result in increased PAH emissions. However, this is largely due to the efficiency of the stoves and the incomplete combustion of the organic biomass material. When biomass is cofired in more efficient coal combustion systems, the PAH emissions are reduced. For example, Chao and others (2008) studied the effect of cofiring rice husks and bamboo with coal on PAH emissions in a laboratory-scale combustion system. The greatest increase in emissions of PAH was caused by decreasing the excess air in the system from 30% to 10%. The increased PAH emissions were therefore due to incomplete combustion and unburnt materials being released. However, when the excess air was stable, the addition of biomass to baseline coal combustion caused a decrease in PAH emissions. At 30% cofiring of rice husks, the PAH emissions were decreased by around 20%. At biomass ratios higher or lower than this, the PAH increased as with the addition or reduction in biomass ratio. When cofiring bamboo, the optimum PAH reduction (up to 15%) was seen at 20–30% bamboo. Chao and others (2008) suggest that the reduction in

Table 4 Trace element content of biomass fired at Drax Power Station, UK (Ghent, 2009)

Element	Milled Palm nut	Soya	Olive hulls	Olive stone	Wood cake	Citrus	Straw pulp	Cereal	Grass	Cocoa	IPC* Bean husk
Arsenic, mg/kg	4.75	0.1	<1	<1	<1	<0.1	1	<0.05	1.5	0.1	5
Cadmium, mg/kg	0.4	0.09	<1	<1	<1	<0.02	<0.5	<0.08	0.5	0.3	3
Chromium, mg/kg	15.5	2.5	2	15	<5	2.6	8	1.5	2	4	30
Copper, mg/kg	25.5	3.8	2	11	2	6.7	7	25	2	45	50
Fluorine, mg/kg	50	14	<5	5	<120	nd	–	–	–	<5	–
Lead, mg/kg	15	0.8	<1	<1	<1	1.8	5	3	3	1.9	20
Mercury, mg/kg	0.1	0.01	<0.01	<1	<0.01	<0.01	<0.05	<0.05	–	0.03	0.1
Nickel, mg/kg	5	0.9	<1	8	1	1.3	3.5	0.05	2	8	30
Vanadium, mg/kg	10.5	nd	<1	2	1	1.2	–	1.0	0.3	1.6	20
Zinc, mg/kg	44	65.1	6.9	15.6	<50	nd	15	65	14	75	80
Moisture, %	4.6	12.4	11.02	15.6	8.3	9.3	12	12	10	12	–
Ash, % wt/wt dry	5.95	4.1	0.3	7.7	2.4	5.7	10	5	5	8	20
Sulphur, % wt/wt dry	0.25	0.07	0.03	0.08	0.03	0.07	0.2	0.2	0.08	0.18	0.4
Chlorine, % wt/wt dry	0.13	0.01	0.01	0.13	0.01	0.05	var.	0.1	0.1	0.01	0.4
CV†, gross MJ/kg	17.25	15.3	18.2	16.2	18.6	15.4	15.5	20.4	16–19	15.2	–

*IPC Integrated Pollution Control – limits set by the UK Environment Agency which are specific to Drax Power Station
†CV calorific value
nd non-detect

PAH emissions with biomass co-combustion is due to the increase in relative residence time. Biomass, with a high concentration of volatile matter, burns considerably faster than coal. Biomass cofiring with coal therefore leads to more efficient destruction of organic species and a lower PAH emission rate.

Fitzpatrick and others (2009) fired coal and biomass briquettes in a 30 kW boiler and noted that the presence of the coal ‘significantly’ suppressed the formation of PAH and phenols. It was also noticed that the briquetting of the fuel helped to reduce the emissions of PAH from biomass combustion. Although Fitzpatrick and others (2009) gave an excellent explanation of the formation of PAH species in small-scale coal and biomass cofiring, it is not considered relevant here. It is well established that the efficient combustion of coal in full-scale plants does not cause PAH emissions in any great quantity (Sloss, 2001) and that the cofiring of biomass with coal will increase the combustion efficiency of the biomass in such a way as to reduce PAH emissions significantly.

3.1.6 Dioxins and furans

Organic emissions are commonly the result of inefficient combustion and, at the temperatures encountered in most

large-scale coal-fired plants, organic emissions are not considered an issue. However, the presence of Cl in biomass materials could, theoretically, lead to an increase in dioxin (polychlorinated dibenzo dioxin; PCDD) and furan (polychlorinated dibenzo furan; PCDF) emissions. Although the boiler temperature would be too high for these species to be produced or released from the boiler, there is the possibility of de novo synthesis (synthesis from smaller precursor species) of these compounds in cooler downstream areas of the plant, such as the ESP (electrostatic precipitator). However, there does not seem to have been anything published on this which would imply that it has been encountered much in practice. Leckner (2007) agrees that there appears to be some limited information which would suggest that cofiring, if anything, reduces dioxin and furan emissions but that more research is needed.

A previous IEA CCC report by Davidson (2005) on chlorine in coal and cofiring systems suggested that the addition of coal to MSW combustion systems is actually an effective method of suppressing dioxin and furan formation. Cofiring coal at 16% (by weight) reduced PCDD/PCDF emissions by 95%. A similar effect was noted when cofiring Greek lignite with sewage sludge where emissions from firing the sludge alone reached 300 ng/kg (toxic equivalent values) compared to 74–158 ng/kg when firing the sludge (10% by weight) with the lignite.

3.1.7 Fine particulates (PM_{10/2.5})

Biomass combustion alone is considered to be such a potentially significant source of PM₁₀ (particles with an aerodynamic diameter below 10 µm) emissions that the IEA Bioenergy Task 32 on Biomass Combustion and Cofiring was established to promote the implementation of technical measures for particulate matter reduction. The countries involved are Austria, Denmark, Germany, the Netherlands, Norway, Sweden and Switzerland. The study has so far concentrated on determining the most appropriate methods for particulate measurement and monitoring (Nussbaumer and others, 2008).

The measurement of particles, especially fine particles PM₁₀ and PM_{2.5} is a significant challenge and has been dealt with in previous reports from the IEA CCC (Sloss, 1998, 2004). The Task 32 study has concentrated largely on emissions from small-scale domestic stoves, open fireplaces and industrial stoves. Particle emission factors for manual wood combustion devices range from less than 20 mg/MJ under ideal conditions to over 5000 mg/MJ under poor conditions (Nussbaumer and others, 2008). The study did not compare emissions from large-scale biomass combustion or full-scale coal and biomass co-combustion plants. It is likely that the larger scale of these systems, the more efficient combustion conditions and the application of particulate control systems such as ESP and fabric filters will ensure that particulate emissions from large-scale combustion of biomass are significantly lower than those from the more inefficient smaller systems.

Particulate emissions correlate with the incombustible ash content of the fuel. De Wilde and others (2007) quote a PM₁₀ emission factor for large stand-alone biomass power plants of 5 g/GJ. The emission factor for cofiring biomass in a coal-fired plant is less than half of this, at 2 g/GJ. No

emission factor was listed for coal combustion with no cofiring. Most biomass materials have a lower ash content than coal, although sewage sludge, MBM (meat and bone meal) and poultry litter have a significantly higher ash content than coal. Higher ash and particulates in the combustion system means more work for the particulate control devices. Further, dried sewage sludge may contain higher concentrations of CaO which can neutralise the surface acidity of the particles and reduce ESP efficiency (Fernando, 2007).

Chao and others (2008) note that, during co-combustion of biomass with coal, the ultra fine mode particles are shifted to a larger size range. Rice husks (agricultural residues) and bamboo (from waste scaffolding from Hong Kong) were cofired with coal in a laboratory-scale pulverised fuel combustion test facility at different ratios. The study showed that the PM₁₀ and PM_{2.1} emissions decreased as the ratio of biomass to coal increased, as shown in Figures 9 and 10. PM_{2.1} was measured rather than PM_{2.5} as this was the size cut-off provided by the measurement device used. The figures show that, although the fine particulate emissions were reduced as the ratio of biomass increased from 0% to 30%, above that, the emissions increased and eventually exceeded the baseline emissions for coal alone at 100% biomass. If the excess air ratio was increased, the fine particulate emissions were reduced at all biomass firing ratios.

Closer study of the particle size distribution showed that the peak number concentration (particles/cm³) shifted towards larger diameters as the amount of biomass increased, as shown in Figure 11. With coal alone, the particles at maximum number density were 1.38 µm diameter (baseline) and increased to 2.29 µm with 50% rice husk. A slight shift was also noted when bamboo was used as the biomass and this was reported to be due to coagulation of the particles. Burning the biomass materials alone also shifted the peak

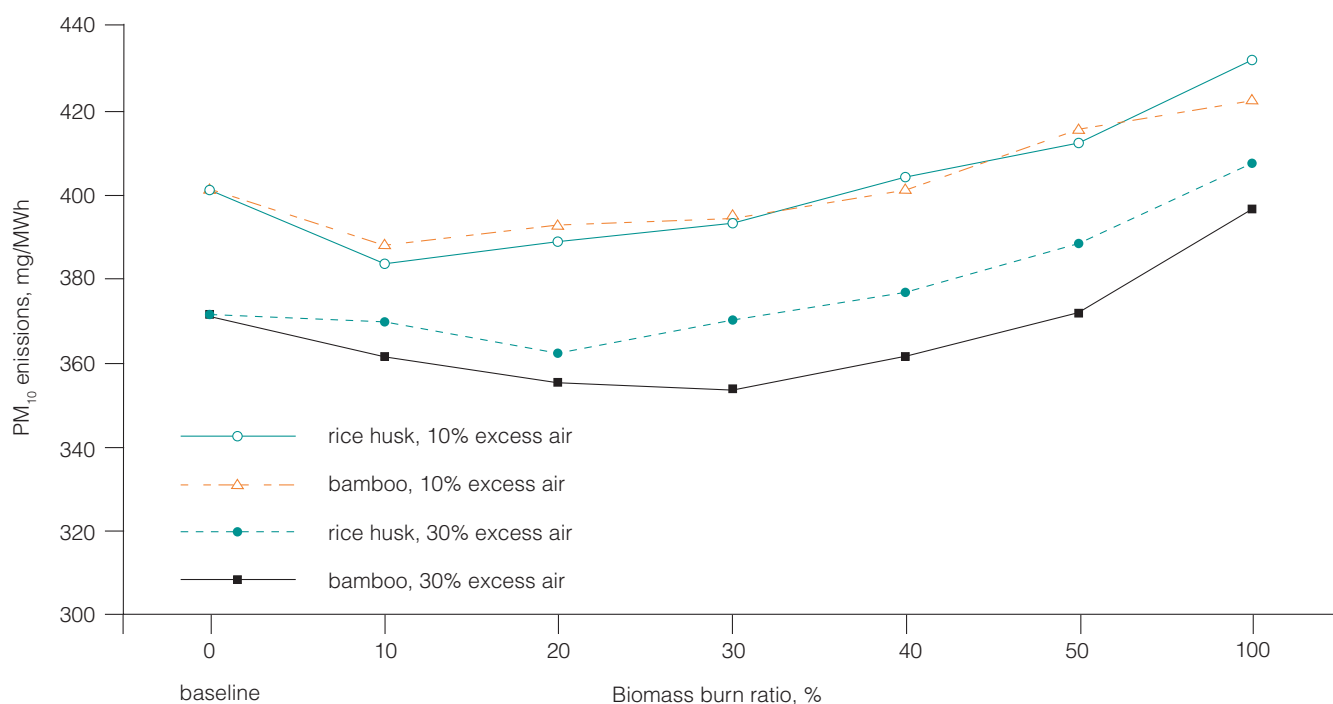


Figure 9 PM₁₀ emissions at various biomass burn ratios and excess air levels (Chao and others, 2008)

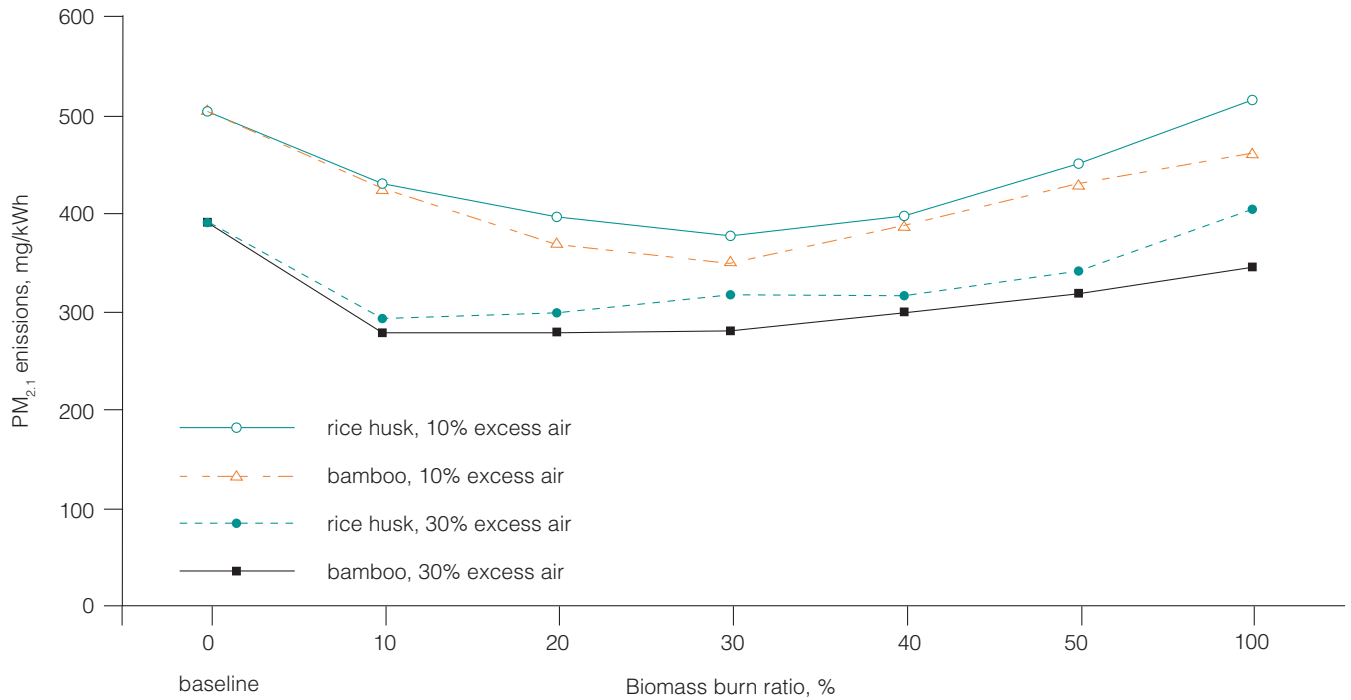


Figure 10 $PM_{2.1}$ emissions at various biomass burn ratios and excess air levels (Chao and others, 2008)

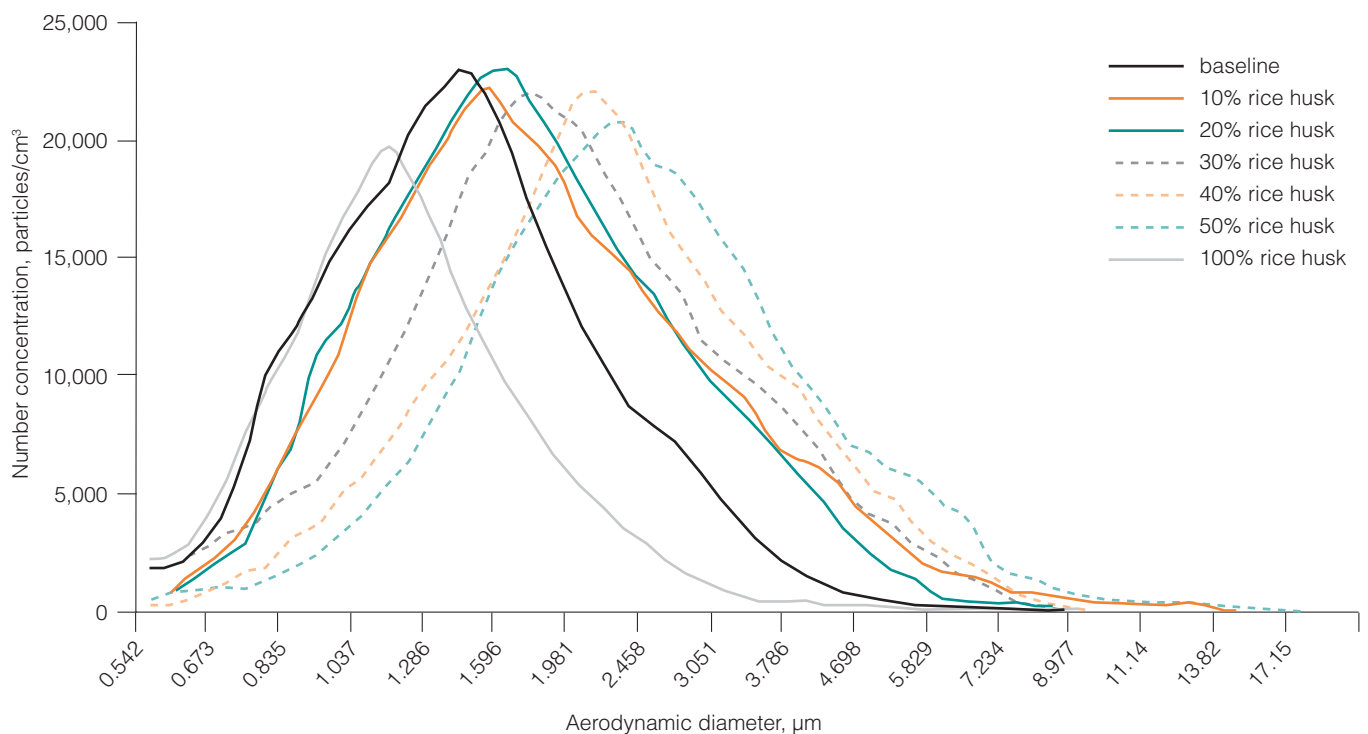


Figure 11 Particulate matter number concentration from rice husk with different biomass burn rates at 30% excess air (Chao and others, 2008)

particle sizes back down to around $1 \mu\text{m}$ implying that it was a synergistic effect of the co-combustion of the coal and biomass which led to the increase in average particle diameter. Biomass combustion alone was also reported to produce more ultra fine ($<PM_{1}$) particles than coal combustion alone. Chao and others (2008) suggest that the increase in ultra fine mode particles from biomass combustion is due to the increased concentration of Cl, Na, K, Mg and Ca in the systems and the presence of a high concentration of

condensable aerosols during biomass combustion. The ultra fine particles formed through nucleation mode and by gas-to-particle conversion processes.

Another consideration with respect to $PM_{10/2.5}$ emissions is the formation of secondary particles in the cooling flue gas and plume. These secondary particles are commonly formed by the condensation and reaction of gaseous SO_2 and NO_x with species such as ammonia in the air. Since emissions of

SO₂ and NO_x are generally reduced in biomass co-combustion systems, as discussed earlier, it can be assumed that the formation of secondary PM_{10/2.5} would also be reduced.

3.1.8 Ash

A previous report by IEA CCC (Sloss, 2007) reviewed the issue of trace elements in fly ash – measurement, legislation, the effect on by-product use and beneficiation options. The interested reader is referred to this 2007 report for more detailed information. Another IEA CCC report (Couch, 2006) deals with ash management in coal-fired plants but does not cover the effects of biomass cofiring.

Some biomass materials, especially wood, have very little ash content (1% or less) which can lead to less waste ash (Eisenstat and others, 2009).

As stated by Vamvuka and others (2009), the properties of ash material formed during the combustion of a fuel mix cannot be predicted from the known characteristics of the ash formed from each fuel. The interactions between the ashes from different fuels are largely unpredictable. Biomass fuels tend to have reactive ashes due to the high content of alkali metals. These species may interact with fuel gases, especially sulphur and chlorine, to form new compounds during combustion.

The unburnt carbon content of fly ash is reported to be reduced when 10–20% biomass is introduced to coal firing. However, at higher biomass firing rates there may be an increase in unburnt carbon due to the low ash content contribution from the biomass (Boneham, 2008).

The trace elements of most concern with respect to fly ash utilisation are generally chromium, especially the cytotoxic and carcinogenic Cr(VI) form, and radioactive elements such as ²²⁶Ra, ²³²Th and ⁴⁰K. Acceptable limits for Cr(VI) in cement and concrete have been set by the European Commission (EC). Based on measuring trace element behaviour at pilot- and full-scale plants, Van Eijk and others (2009) suggest that Cr (probably in the form of Cr VI) is more available for leaching in coal ash than in ash produced from cofiring coal with wood pellets (40% on an energy basis).

At the moment, radioactivity in building materials is only legislated for in Israel. This is not because the coal in the area is any more reactive than other coals but rather appears to be due to a national concern in Israel over the potential threat from the radioactive content of all building materials resulting in limits that are far more stringent than those seen elsewhere (Sloss, 2007).

The EU TOMORED project demonstrated that the cofiring of 10% sewage sludge with coal could almost triple (1.2 mg/kg up to 4.2 mg/kg) the concentration of Cd in the ESP ash at the test boiler at the University of Stuttgart (Thorwarth, 2006). However, in most situations it would appear that any trace element concentration increase in the ash due to cofiring is minor and does not tend to cause any problems with ash sales or with ash leaching. For example, Lamers and others (2000) note the following effects on ash:

10% petcokes	trace elements in the leachates from the ash were well below detection limits for most elements, although Mo and Ni were slightly elevated in the fly ash itself;
<6% sewage sludge	slightly elevated concentrations of Cd, Pb, and Zn in the leachate;
10% paper sludge	concentrations of Cd, Hg and Zn increased by a factor of around 1.5 in the ash leachate;
<3.6% wood	no detectable increase in the leaching of trace elements from the fly ash.

Achterbosch and others (2005) studied the effect of using trace element laden materials such as iron works waste, basalt and slag along with coal in cement and concrete production and found, unsurprisingly, significant increases in species such as Co, Cr, V and Zn.

Sewage sludge generally contains higher trace element content than coal and this is reflected in an increase in the concentration of some trace elements in the ash. Emissions of Hg and Cd can increase when sewage sludge is cofired with coal but the presence of sulphur can cause an increase of Hg capture in the ash. Concentrations of As, Pb and Se in the ash increased in several studies reviewed by Barbosa and others (2009). Barbosa and others tested the effect of sewage sludge cofiring on ash in a pilot-scale (0.3 m x 0.3 m x 5 m) bubbling FBC (BFBC) system in Portugal. The sewage sludge used had been stabilised to form a soil conditioner known as Biogran® and this was cofired with Colombian El Cerrejón coal. A comparison of the ultimate analysis of each fuel is shown in Table 5. Table 6 then shows the characteristics of the ash at different stages of the particulate control system when firing

Element, wt%, dry basis	Bituminous coal	Biogran®
C	79.1	30.9
H	5.0	3.8
N	1.8	3.7
Cl	0.06	0.07
F	<0.01	<0.01
S	2.15	0.96
P	0.51	3.11
Ca	0.20	5.3
K	0.04	0.6
Na	0.03	0.2
Mg	0.02	0.5
Si	1.8	7.1

Table 6 Bulk characterisation of the bottom and fly ashes collected during combustion of coal and co-combustion of coal and Biogran® in a BFBC unit (Barbosa and others, 2009)

Element	Combustion test A – coal alone			Combustion test B – coal + Biogran®		
	bottom ash	first cyclone ash	second cyclone ash	bottom ash	first cyclone ash	second cyclone ash
As	<0.78	6.2	2.9	3.1	5.0	5.3
Cd	<7.8	<15	<7.2	<7.9	<12	<11
Cr	21	409	151	159	466	336
Cu	<10	71	72	123	329	473
Hg	7.4	0.84	2.4	5.0	7.4	4.8
Ni	<15	179	131	97	282	305
Pb	<24	<47	<23	64	251	360
Zn	73	338	362	413	1,211	1,583
Fe	2,462	23,952	21,405	4,871	26,125	31,754
Al	6,333	34,434	38,088	6,067	33,607	47,784

Table 7 Ash legislation relevant to biomass cofiring (Fernando, 2007)

Standard/legislation	Details
Original German standard (superceded by EN450-2)	Fly ash from cofiring sewage sludge was allowed provided that the input of sewage sludge, on a mass basis, did not exceed 5% and that the input concentration of prescribed trace elements in the sludge did not exceed a given limit
European Standard EN450-1 (original version)	Only fly ash from pure coal or anthracite combustion to be used for cement or concrete
European Standard EN450-2 (updated since 2005)	Ash from cofiring can be used as long as the per cent of secondary fuel does not exceed 20% by mass of the total fuel and if the derived amount of ash from the co-combustion material is not greater than 10%
USA ASTM C618	Fly ash for use in concrete should be entirely from coal combustion, although exceptions are allowed depending on industry requirements

the coal alone and when cofiring coal and Biogran® at 1:1 (weight basis). The results show that the majority of the trace elements are collected in the first and second cyclones, with slightly more in the second cyclone. Clearly the co-combustion of the Biogran® leads to higher ash concentrations of all the elements measured, especially Cr, Cu, Ni, Pb, Zn and Fe. These species were consistently more concentrated in the fly ash, especially the second cyclone, than in the bottom ash.

The ashes from the study were subjected to a standard leaching test (EN 12457-2) to determine the mobility and leaching potential of the species captured in the ashes. The toxicity equivalent was also evaluated along with the effects on micro-crustacean and bacteria species. Cr and Cr(VI) were found in the first cyclone fly ashes of both the coal combustion alone and the cofiring with Biogran® and in the second cyclone of the cofiring test. The leachate from the

cofiring test had the highest toxicity level although the actual ecotoxic levels were still 'low'. However, the toxicity levels measured would mean that, under French Regulations, the bottom ashes from both the combustion of coal alone and the cofiring test would be defined as non-toxic residues whereas the fly ash from both tests would be classified as toxic.

Although it is clear that the chemical characteristics of biomass can change the chemistry of fly ash and increase the concentrations of some trace elements, it would seem that the changes will not always result in a negative effect. However, the main barrier to the use of ash from cofiring is legislative and based on arguably simplistic and prescriptive rules rather than on performance-based criteria which would guarantee the suitability of the ash for use. These are summarised in Table 7. Standards such as that in the EU (EN450) specify that ash to be used in concrete must be derived entirely from coal combustion. This means that any plant wishing to switch

to cofiring biomass must take into account the potential for loss of revenue from fly ash sales and the prospect of dealing with a new waste stream. It could be argued that the general ban on the use of fly ash for co-combustion is too strict and could mean that fly ashes which are fit for use are being wasted. Extensive testing in the Netherlands has shown that fly ash produced from cofiring relatively high percentages of biomass with coal can still produce fly ash that meets the necessary performance standards (Fernando, 2007). The standard in the USA (American Standards for Testing and Materials, ASTM C618) is more flexible in that the US EPA and individual states can determine which ash materials can be used. It is also the case that the cement and concrete industry in the USA have their own specifications and preferences for the materials used and these are likely to be more performance based – that is, they are likely to accept the most suitable materials for purpose. The US EPA currently has a strategic plan to promote the use of fly ash and other ‘products’ from coal combustion with a target of 50% recycling to be achieved by 2011 (US EPA, 2010).

Koppejan and Baxter (2005) investigated the impact of cofiring on fly ash and its applicability in concrete production. The study covered both Class C (subbituminous) and Class F (bituminous) fly ashes and compared these with fly ashes containing 0–40% biomass derived material. In each case, the fly ash was used to replace 2% of the cement. The concrete samples were prepared and aerating agent (surfactant) added to each to reach the required ASTM compliant air entrainment levels. Figure 12 shows the amounts of agent required for each type of fly ash studied. Air entrainment is essential in concrete to prevent failure during freeze-thaw cycles. The amount of aerating agent required increases quite significantly with the amount of biomass ash included, although the increased use of the aerating agent is not expensive. However, it is not obvious during the cement production that the aerating agent is required. Concrete could be produced from biomass containing ash but it could fail quite dramatically under freeze-thaw conditions. It is therefore important that aerating conditions are monitored if biomass cofired ash is to be used for concrete production. Figure 13 shows the impact of biomass ash on flexural strength. The effect is relatively minimal and is, in the long term (over two months) actually beneficial. Another effect of the biomass ash presence is the delay in set time by around 2–4 hours.

Koppejan and Baxter (2005) concluded that, despite the requirement for increased aerating agent, the difference in ash from cofired systems to that from coal alone is minimal and manageable.

In a previous IEA CCC report, Fernando (2005) reviewed several papers which suggested that the EN450 standard, which only allows fly ash from coal combustion alone to be used in concrete (similarly to the ASTM standard discussed above), was too strict. Studies with ash produced from cofiring coal with straw up to 14% and wood up to 47% produced ash that complied with all the requirements of EN450 except for the limit on carbon in ash (7% for cofired wood ash compared to 5% set in EN450). These elevated carbon-in-ash contents could be avoided by using higher excess air levels during combustion. Despite the

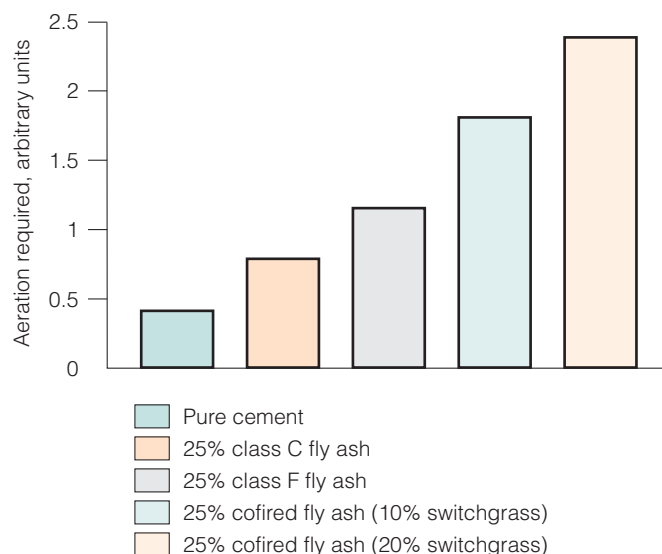


Figure 12 Required amount of aerating agent required to generate air entrainment within ASTM specifications for a variety of fly ash compositions (Koppejan and Baxter, 2005)

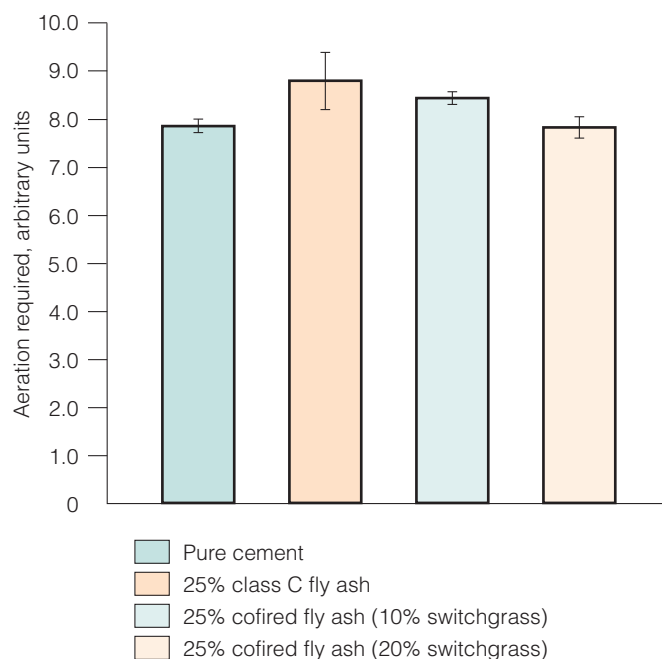


Figure 13 Concrete flexural strength and its dependence on fly ash composition (Koppejan and Baxter, 2005)

carbon-in-ash issue, the ashes from co-combustion tended to produce concretes which showed higher compressive strengths than those produced from coal alone.

3.2 Grate-fired systems

As shown in Figure 2 (page 10), around 6% of the plants cofiring biomass in 2005 were grate-fired systems. A previous report from IEA CCC (Fernando, 2007) dealt briefly with biomass use in grate- or stoker-fired systems. These systems vary in the way the fuel is delivered to the boiler. Most systems feed the fuel in from above but a few

have a system whereby the fuel is fed in from below by a grate, while air is blown through the bed of the fuel. These types of combustors can fire a wide range of fuel made up of larger pieces than in pulverised coal fired systems and they can also cope with relatively moist fuels. As a result, grate or stoker boilers tend to be less efficient than pulverised fuel or fluidised bed systems and therefore the emissions can be significantly higher. Most grate or stoker systems are fitted with some form of particulate controls, such as cyclones, ESP, baghouses or wet scrubber systems. Stoker boilers, especially those with travelling grates, are suitable for firing tyre-derived fuel with coal. The 74 MW Jennison Plant in New York, USA, cofires up to 4.5 million tyres (around 25% of the thermal input of the plant) annually with coal. Following the addition of the tyre material to the fuel mix, there were no detrimental effects on emissions and SO₂ emissions actually decreased slightly. There was an increase in zinc in the fly ash and an increase in other metals in the bottom ash. Magnetic separation equipment was installed in the ash pond to remove metals from the bottom ash (Fernando, 2007). No further published material was found relating to any detrimental effects of biomass co-combustion in stoker- and grate-fired systems.

3.3 Fluidised bed combustion

Fluidised bed combustion (FBC) boilers are much more suitable to cofiring and multi-fuel use and allow a much higher ratio of biomass to coal or peat use than in pulverised systems (Kangas and others, 2009). Scandinavia has around 150 FBC boilers firing or cofiring biomass (Fernando, 2005).

Control of fuel particle size and ash sintering is an issue with respect to plant performance and so woody biomass (ash melting point >1000°C) is better suited to FBC combustion than herbaceous material (such as straw, with an ash melting point of <700°C).

Dai and others (2008) reviewed numerous papers on biomass cofiring and found that, in most cases, cofiring in FBC systems reduced the CO₂, SO₂ and NO_x emissions and that the emissions decreased with increasing wood to coal ratio. Higher straw to coal ratios cause an increase in the HCl concentration.

Leckner (2007) has studied the effect of different fuel ratios on emissions from the Chalmers 12 MW circulating FBC (CFBC) boiler in Sweden. The results are summarised in Figure 14. As expected, the lower sulphur content of wood chips and sawdust resulted in lower SO₂ emissions at higher biomass ratios. Although the nitrogen concentration is lower in the biomass than in the coal, it was noted that the NO emission from pure biomass is higher than that from pure coal. Leckner (2007) explains that this is due to the capability of char to reduce NO. Since the char content of wood combustion is low, the NO is not oxidised as it is when coal is present. As coal is added to biomass combustion, the char concentration builds up until it reaches a level where NO can be reduced. It can therefore be argued that cofiring coal with biomass materials can be beneficial in reducing NO emissions from biomass combustion alone and this has been confirmed in studies of sewage sludge combustion (Leckner, 2007). In

situations such as this, the addition of coal to an existing biomass combustion system can be beneficial in enhancing the efficiency of biomass combustion.

N₂O is not commonly emitted in significant concentrations from pulverised coal combustion but can be a problem with FBC systems. As was shown in Figure 14, the N₂O emissions decreased with increasing biomass ratio as the concentration of nitrogen in the biomass is lower than that in coal (by an order of magnitude). Leckner (2007) suggests that the nitrogen is mostly transformed into NH₃ which is not an important precursor of N₂O at the temperatures encountered. The high concentration of hydrogen from biomass combustion will also play a role in N₂O reduction. The light weight of the sawdust means that it is carried with the gas to the top of the burn area, increasing the temperature in that area and destroying N₂O. This high temperature destruction of N₂O is what ensures N₂O emissions from pulverised systems are not an issue.

A previous study at the same CFBC plant by Amand and others (2001) had shown that up to 50% of the coal (energy basis) could be replaced by dry sewage sludge without any problems at the plant. The dried sludge had a residual water content of about 20% and the ultimate analysis of the sludge showed the oxygen content to be high (30%). The nitrogen content of the sewage sludge was high – 7.10 wt% compared to 1.70 wt% in the coal. Although the NO production in the boiler was noted to be high, the reduction through the plant was significant and the resulting emissions of NO_x were still low. N₂O emissions were also low. The sulphur content of the sludge (1.90 wt%) was also greater than that in the coal (0.90 wt%).

According to Kuprianov and others (2009), the combustion of biomass material such as rice husks in FBC systems can cause ‘substantial’ NO_x emissions (120–180 ppm) and they suggested that, when this occurs, it is due to insufficient primary air mixing. A laboratory-scale innovative swirling FBC (SFBC) was developed using an annular spiral distributor as the swirl generator for the primary air. The rice husks were fired in the ‘wet’ form (8–35% moisture) and secondary air was injected tangentially to reduce CO. It was noticed that the NO emissions could be reduced through moisturising of the as-received rice husks. This is suggested to be due to the reduction of NO by CO and the optimisation of the excess air. Kuprianov and others (2009) conclude that the emissions of NO and CO from FBC combustion of biomass such as rice husks can be minimised by optimising the moisture content to 20–25% and the excess air to 40–50%. This treatment can control CO emissions to under 350 ppm and NO to 130–140 ppm (6% O₂, dry flue gas basis) while the combustion efficiency was as high as 99.4–99.6%.

Wan and others (2008) discuss the cofiring of coal and MSW in a CFBC boiler. Studies were carried out on various concentrations of RDF-5 (densified refuse derived fuel) with coal in a 103 MWth, 27 MW electric co-generation CFBC boiler in Taiwan. SO_x emissions decreased slightly from around 190 ppm at 0% (heat input) RDF-5 firing with coal down to around 175 ppm at 30% RDF. NO_x emissions also decreased from over 80 ppm at 0% RDF-5 to 70 ppm at 30%

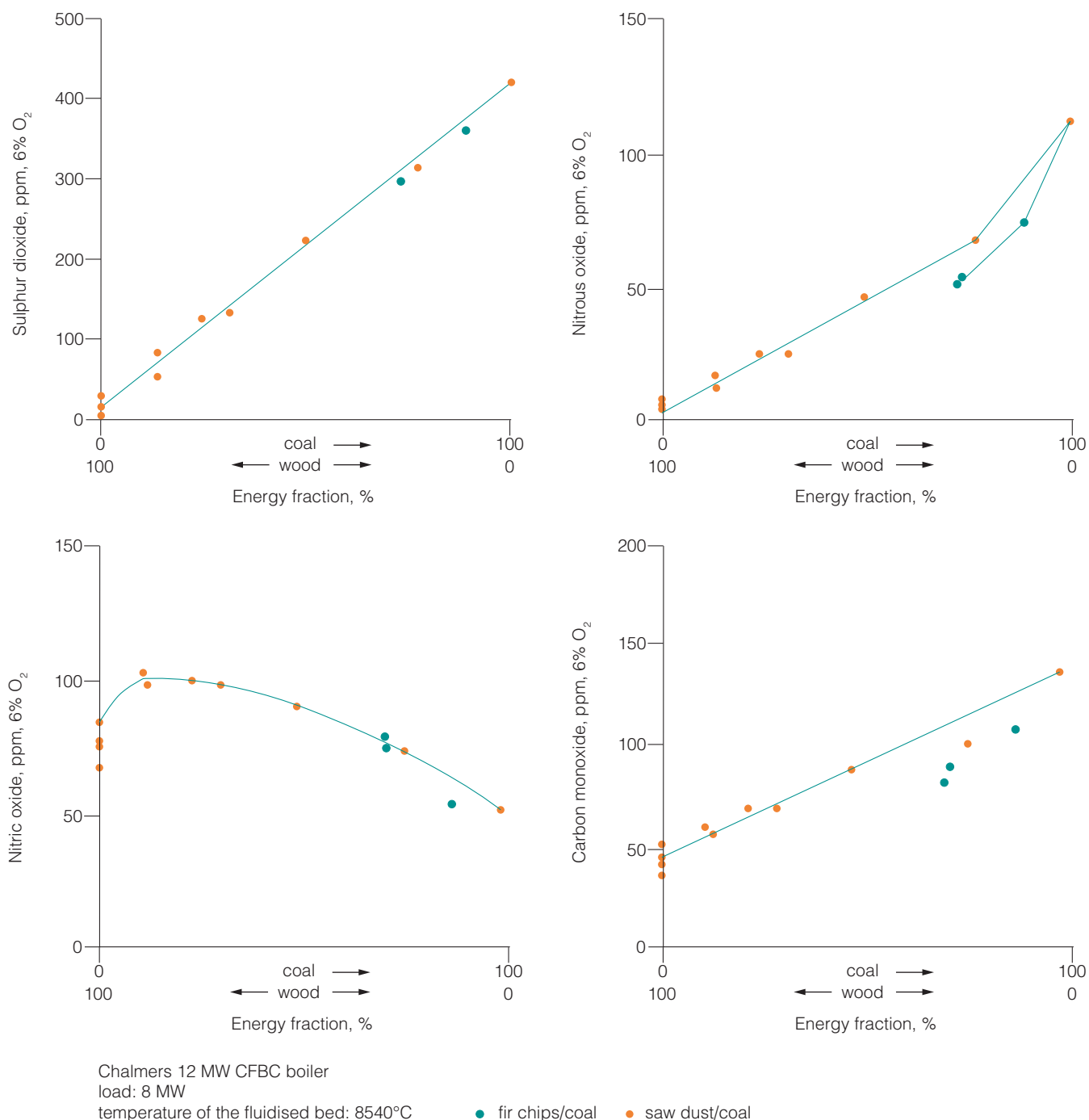


Figure 14 Emissions from co-combustion of coal and wood chips in the Chalmers FBC boiler (Leckner, 2007)

RDF-5. As the proportion of MSW increased, so did the HCl emissions. The addition of CaO could be used to control HCl emissions, and maintaining combustion at around 650°C also maximised HCl capture in the ash. With the lime treatment, emissions of HCl from the plant were below 1 ppm. Dioxin production during the cofiring of MSW in the CFB system was also an issue. With coal combustion alone the dioxin emissions were around 0.01 ng-TEQ/m³ and this doubled to over 0.02 ng-TEQ/m³ at 30% cofiring of RDF-5 (% in total heat input). The presence of sulphur in the combustion system can restrict the formation of dioxins due to interactions which inhibit the catalytic effect of metals in the flue gas. Further, the presence of lime in the bed helps to trap the Cl and prevent the formation of dioxins.

Cao and others (2008) studied mercury emissions during the

cofiring of subbituminous coal (Powder River Basin) and biomass (chicken waste, wood, coffee residues and tobacco stalks) in a laboratory-scale FBC system. The mercury content of the coal was around 0.12 ppm compared to concentrations of 0.01 ppm or below for all of the biomass materials. Figure 15 shows the variation in mercury emissions during cofiring of the coal with the various types of biomass. The graph shows the difference between the mercury concentration in the original fuel compared with the eventual mercury emissions from the system. Since mercury emissions are notoriously difficult to measure accurately, Cao and others (2008) have used two different monitoring methods:

- SCEM – a PS® Analytical Continuous Emissions Monitoring (CEM) system;
- OHM – the Ontario Hydro method, a wet chemical method based on impinger solutions.

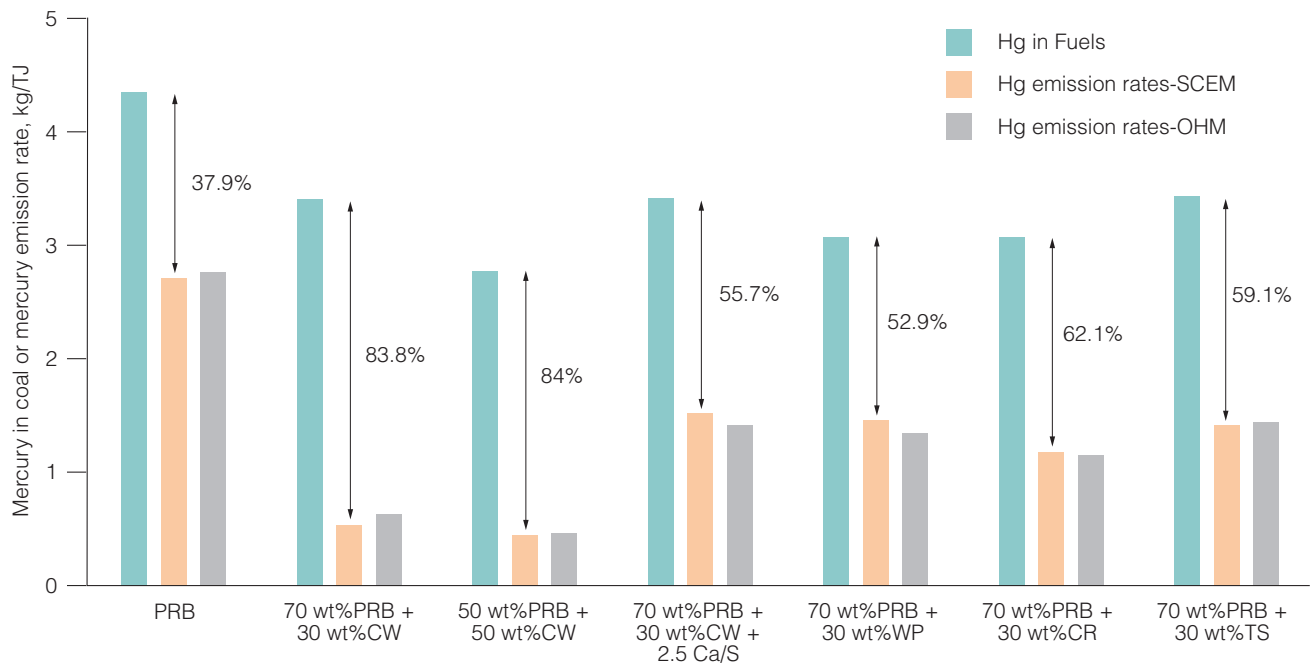


Figure 15 Variation of mercury emission during cofiring of subbituminous coal and biomass (Cao and others, 2008)

When firing the coal alone, around 38% of the mercury was captured in the existing pollution control system (quartz filters). Cofiring high chlorine fuels such as chicken waste (22,340 ppm, Cl, by weight) could reduce mercury emissions by over 80% whereas low chlorine fuels such as wood pellets (132 ppm) only reduced mercury emissions by 50%. However, although tobacco stalks had a high chlorine content (4237 ppm) the co-combustion of this biomass material in the FBC system did not reduce mercury emissions as significantly as might have been expected for such a high chlorine content. So, although the mercury emissions were strongly correlated to the gaseous Cl concentration, they were not necessarily correlated to the chlorine content of the fuels.

This means that the behaviour of mercury and chlorine is more dependent on the chemistry of these species in the coal and the conditions of combustion than the actual concentrations themselves. The chemistry of mercury and chlorine in coal combustion is discussed in more detail in several IEA CCC reports (Sloss, 2002; Davidson, 2005).

Figure 16 shows the speciation of the mercury during the different cofiring variations. It is clear that the higher the fraction of mercury in the elemental state in Figure 16, the lower the mercury emission in Figure 15. The addition of limestone to the FBC system during cofiring was found to increase mercury emissions to the level expected when firing

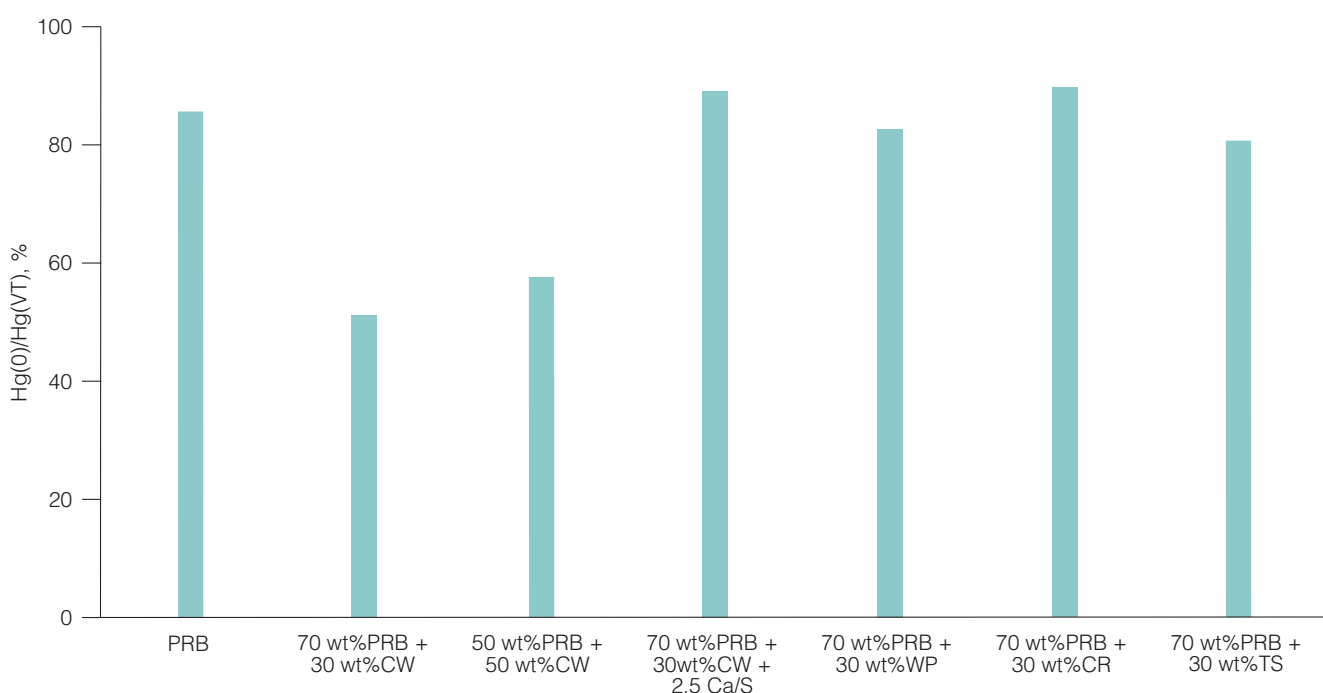


Figure 16 Mercury speciation during cofiring of subbituminous coal and biomass (Cao and others, 2008)

coal alone. It seems the limestone eliminated the chlorine in the flue gas thus reducing the oxidation effect. This means that much of the mercury remained in the elemental state and was not captured in the pollution control systems.

Cofiring straw or sewage sludge in a pressurised FBC system (PFBC) has been reported to slightly increase NO_x emissions. This appears to be dependent on the blending ratio, fuel properties and combustion conditions (Dai and others, 2008).

3.4 Gasification

Gasification of coal requires temperatures of at least 900°C. Biomass needs anything from 800°C upwards. Since biomass is a more reactive fuel than coal, the combustion can be quite different. Fernando (2009) discusses the behaviour of biomass in co-gasification systems with coal. The design of gasification systems is quite distinct and the addition of biomass as a new fuel to existing plants could require alterations to the fuel delivery systems. This is outwith the scope of this report but is dealt with in several IEA CCC reports (Fernando, 2005, 2007, 2009). There is little published data on changes in emissions from gasification systems cofiring biomass materials and so this Section only includes a few brief examples of the types of changes seen in gasification systems if biomass fuels are added as co-combustion fuel.

The presence of metals in the syngas of gasification systems is undesirable as they can cause corrosive effects. Biomass ashes containing high levels of alkali metals can increase fouling and cause detrimental ash leaching characteristics (Fernando, 2009).

Al-Kassir and others (2009) studied the gasification of waste materials from cork processing (sawdust, sandpaper dust and triturated wood) in an experimental-scale gasifier in Portugal. The biomass fuel showed high volatile matter content and low S and Cl, which would suggest that slagging and fouling for this material may not be as much of a problem as for other biomass materials.

Coal and RDF (50% wood, 15% plastics, 35% paper and cardboard) were gasified in a laboratory-scale bubbling fluidised bed gasifier. Although the RDF material contained HCl, no increase in HCl was measured during cofiring. It appeared that around 22% of the chlorine within the RDF material was converted to HCl and that all of this was retained with the condensates. High temperatures favoured the reduction of the formation of H₂S, NH₃ and HCl. Steam promoted the retention of S and N in the solids whilst O₂ favoured the volatilisation of these elements into the gas phase. ZnO was found to be an excellent catalyst for the agglomeration of H₂S into the bed (Gulyurtlu and others, 2007).

Pinto and others (2006) studied the co-gasification of biomass (pine), sewage sludge and edible oil wastes with coal in a bench scale fluidised bed gasifier in Portugal. The system coped well with biomass mixtures up to 60% w/w with the coal but the use of edible oil waste had to be kept below 10%

due to the effect on the fuel flow behaviour. The gasification mixture from co-gasification had higher hydrocarbon contents than from coal alone. The most problematic gases were, again, NH₃ and H₂S with sewage sludge causing the greatest increase in NH₃. It was reported that 'most' of the heavy metals present were trapped in the char. However, this char is free of Hg and the metal leachability is undetectable so utilisation is still possible.

Potentially mercury is the element of most concern as it may be high in some biomass materials. Mercury remains in the gas phase through gasifiers and is not caught in the char. Mercury may also pass several types of gasifier gas cleaning processes. VTT in Finland is currently studying mercury removal by different sorbents as further improvement of dry gas cleaning methods (Nieminen and others, 2004).

3.5 Comments

From the literature reviewed in this Chapter it would seem that, in the majority of cases, the co-combustion of biomass or waste with coal results in emissions of particulates, SO₂ and NO_x at similar or lower levels than those encountered when firing these fuels alone. That is, the cofiring of these two fuels is often beneficial.

There are, of course, exceptions to this rule due to the high variability of biomass materials. For example, waste tyres can have higher S contents than coal and could increase emissions from a coal plant, although this is likely to be minimal or at least at a level that could be handled by existing pollution control systems such as FGD.

The behaviour of NO_x in combustion systems is complex, as the emissions are related to both the fuel nitrogen and the nitrogen in the combustion air. Although the nitrogen content of biomass is generally lower than that of coal, the emissions from biomass combustion alone are often higher due to the formation of greater quantities of thermal NO_x. However, cofiring biomass with coal in existing pulverised coal combustion systems seems to have intrinsic NO_x reduction effects and there were no examples found in the literature of biomass co-combustion being anything less than beneficial in this respect. In fact, the cofiring of biomass can be handled in such a way as to act as a NO_x reduction strategy for coal combustion systems. This is also true for FBC systems where the addition of coal to existing biomass combustion systems can reduce the NO_x emissions significantly – the greater the proportion of coal, the lower the NO production. Coal combustion alone in FBC systems can cause elevated N₂O emissions. However, the addition of biomass changes the combustion conditions so that more of the nitrogen is transformed into NH₃ which is not a precursor of N₂O, thus reducing overall N₂O emissions. There can therefore be a balance of the proportion of biomass and coal in pulverised coal combustion systems and in FBC systems which can be optimised to reduce NO_x emissions. It is likely that this optimum balance will vary between units and fuel types and should be determined on a case-by-case basis.

The halogen contents of biomass can cause issues with

respect to slagging and fouling but no evidence has been found of any negative effect with respect to emissions. However, the chlorine content of biomass can have an effect on the behaviour of trace elements, especially mercury, during combustion. The trace element contents of biomass materials are highly variable, and in materials such as sewage sludge the concentrations can be significantly elevated. However, it would seem that cofiring these materials with coal does not lead to greater emissions of these elements but rather higher concentrations in the ash.

The higher efficiency of coal combustion allows higher efficiency of combustion of cofired biomass and this leads to lower PAH emissions. Although it is possible that higher chlorine contents of some biomass could result in higher production of dioxins and furans downstream in ESP systems, this has, so far, not been noted in any of the published literature.

Cofiring can change the size distribution of particles released from the combustion zone, but the overall emissions are generally lower.

Although it would seem that emissions of most if not all pollutant species from co-combustion systems are reduced, this can mean an increase in concentrations of these species in the ash. The elevated levels of trace elements and carbon in the fly ash from co-combustion systems resulted in the tightening of legislation in the EU and USA with respect to the use of this material in cement and concrete manufacture. Previously only ash from coal combustion alone was considered usable. However, numerous studies have shown that, although some modifications may be necessary to ensure the performance characteristics of fly ash from co-combustion, the resulting cement/concrete is as good if not marginally better than that prepared from coal fly ash. It was therefore suggested that the existing legislation on the use of fly ash from coal-combustion was too stringent and could be an unnecessary barrier to the option of co-combustion at plants that obtain revenue from the sale of fly ash and new legislation is currently being developed.

4 Case studies

As mentioned previously, although much has been published relating to the co-combustion of biomass and related fuels with coal, a relatively small proportion of this work relates to the effects on emissions. This Chapter reviews a select few studies which have focused on the changes in emissions encountered when full-scale coal-fired plants start cofiring biomass.

4.1 Drax, UK

Drax is currently the world's largest coal and biomass cofiring project. Drax, in Yorkshire, is the largest coal-fired power station in the UK, comprising six separate units totalling 4000 MWe and providing around 7% of the electricity supplied in the country. All six units are fitted with FGD systems with a minimum of 90% sulphur removal efficiency. The operators plan to construct three further 300 MW biomass-fired generation plants.

Drax has a challenging environmental policy programme, the focus of which is on cofiring as well as energy efficiency improvement. The plant has set a target of 12.5% output from renewables by 2010, saving over 2.5 Mt of CO₂/y. The operating company report that the biggest challenge for this target has been the sourcing of sufficient biomass material.

Table 8 shows the fuel use by type from 2000 to 2008. Although the coal use has increased (along with the plant output over the same period), the use of alternative fuels such as petcoke and biomass has also increased. The heavy oil use is for start-up, combustion support and load carrying periods only. The renewable oil mentioned in Table 8 is 'tall oil' – a biodegradable oil produced as a by-product of wood pulping. Only small amounts were used in recent years because the oil proved to be uneconomic to produce and use. Petroleum coke (petcoke) from the petrochemical industry was tested successfully between 2005 and 2007 as a 20% blend with coal. The trial indicated no negative environmental effects and even indicated possible benefits resulting from reduced particulate emissions. Full commercial burn of petcoke was initiated in 2008 at all six units and is now a normal part of the station operation. Although no negative environmental effects were noted from the introduction of petcoke as a

regular fuel at the plant, off-site monitoring of nickel and vanadium was introduced in the local area in conjunction with the Environment Agency and local councils. So far, no negative effects have been reported.

Different types of biomass have been used at Drax. The total biomass burn at the plant in 2008 was over 412 kt. The volumes of the different biomass fuels used in 2008 are shown in Table 9 along with their source of origin. Wood pellets are by far the most common fuel used to date, with the pellets being sourced and transported from as far away as North America and Russia.

The emissions from Drax over the period, 2000-08, are shown in Table 10. As discussed in Chapter 3, emissions from biomass and coal co-combustion can actually be lower than the emissions from the combustion of these fuels alone. Emissions of all major air pollutants at Drax (SO₂, NO_x and particulates) have been reduced significantly during this period. Particulate emissions did increase between 2007 and 2008 and no explanation was given. However, the increase was not enough to raise emissions to anywhere near the legislated limit. Since all the units at Drax are fitted with FGD and deNO_x systems (boosted overfire air), the emissions are already significantly reduced compared to what they would be in an uncontrolled system. With improved efficiency of the plant and the FGD system, the SO₂ emission rate has been reduced from over 2.0 t/GWh in 2002 to just over 1.0 t/GWh in 2008. In the same period, the existing low NO_x burners at the plant were gradually complemented by the boosted overfire air systems. Over this time the NO_x decreased from over 2.6 t/GWh in 2002 to around 1.4 t/GWh in 2008. The installation of the FGD and DeNO_x systems makes it difficult to determine any potential increase in SO₂ or NO_x emissions during the addition of biomass to the fuel mix. But it is certainly the case that any increase is easily controlled by the new pollution control systems. However, the increase in particulate emissions between 2007 and 2008 is curious and may require further investigation.

In a trial in 2005 based on a 10% biomass/90% coal blend (wt%), it was shown that the concentrations of all the trace elements measured in the biomass were consistently lower than that in the coal and therefore the overall result of the

Table 8 Fuel use at Drax power station, UK, 2000-08, kt (DPL, 2009)

Fuel	2000	2001	2002	2003	2004	2005	2006	2007	2008
Coal	8,500	8,360	7,345	9,783	9,167	9,323	10,197	9,815	9,720
Petroleum coke	0	0	0	0	0	64.05	111.3	140.1	267.9
Heavy fuel oil	37.0	40.0	36.0	20.1	17.1	7.9	19.3	21.9	27.2
Renewable oil	0	0	0	0	26.6	68.87	20.75	9.3	1.46
Biomass	0	0	0	4.94	76.7	161.6	43.33	187.4	412.1

Table 9 Biomass used at Drax in 2008 (DPL, 2009)

Fuel	Origin	Amount, t	Nature of fuel
Wood pellets	Baltics	154,146	Residue
Wood pellets	Portugal	51,382	Residue
Wood pellets	Russia	43,763	Residue
Wood pellets	Canada/USA	22,174	Residue
Wood pellets/sawdust	UK	8,914	Residue
Sunflower husk	Ukraine	74,847	Residue
Peanut shells	USA	42,057	Residue
Cocoa shell	UK	5,688	Residue
Miscanthus	UK	4,868	Energy crop
Grape seed flour	Spain	3,614	Residue
Straw pellets	UK	372	Residue
Willow	UK	258	Energy crop
DDGS*	USA	25	Byproduct
Cork fines	Portugal	30	Residue
Total		412,144	

* DDGS dried distilled grains with solubles – a by-product from bioethanol production

Table 10 Emissions from Drax power station, UK, 2000-08, kt (DPL, 2009)

Emissions	2000	2001	2002	2003	2004	2005	2006	2007	2008
CO ₂	19,000	18,735	16,350	21,642	20,519	20,771*	22,764*	22,503*	22,299*
SO ₂	31	45.7	34.6	44.2	26.23	25.25	20.48	20.5	24.5
NOx	59	58.5	49.6	64.7	57.7	57.3	57.13	53.9	38.25
Particulates	1.5	0.3	0.3	0.3	0.4	0.45	0.48	0.33	0.47

* Data for 2005-08 have been verified under the EU ETS, they include the FGD contribution but not any contribution from biomass

cofiring was to reduce emissions of all trace elements not only into the flue gas but also into the other residues such as fly ash. Table 11 shows the reduction in the environmental impact of the 10% biomass cofiring. The emissions of all trace elements to the air are reduced as well as emissions to other media (such as waste water and FGD water). Whilst some of this reduction will be due to the increased capture of trace elements in the new FGD system (as gypsum) and in the ash, there will also be a reduction due to the lower input of these elements to the combustion system since the concentrations of many of the elements were lower in the biomass materials.

4.2 Fiume Santo, Italy

The 640 MW Fiume Santo plant, operated by Endesa, is situated in Sardinia, Italy. The plant is split into several units, only two of which (units 3 and 4; 160 MW each) fire coal. The plants are fitted with ESP, wet FGD for SO₂ control and

Table 11 Reduction in environmental impact from burning biomass at Drax in 2005 (DPL, 2009)

Species	Reduction in emissions to air	Reduction in emissions to other media
CO ₂	145,716 t	–
As	477 g	558 kg
Cd	307 g	7.42 kg
Cr	3339 g	2520 kg
Cu	1522 g	1355 kg
Pb	11690 g	975 kg
Hg	969 g	8.53 kg
Ni	2230 g	2929 kg
V	317 g	5427 kg
Zn	488 g	640 kg

Table 12 Characteristics of coals fired at the Fiume Santo plant, Italy (Conti and others, 2008)

Proximate analysis	Indonesian coal	South African coal
Moisture, wt% (ar*)	9.59	6.82
Ash, wt% (ar)	11.11	12.88
Volatile matter, wt% (ar)	38.87	24.69
Fixed carbon, wt% (ar)	40.43	55.61
LHV, MJ/kg (ar)	25.2	25.8
Ultimate analysis		
C, wt% (ar)	66.46	66.96
N, wt% (ar)	0.99	1.58
H, wt% (ar)	4.82	3.58
O, wt% (ar)	11.37	27.88
S, wt% (ar)	0.89	0.57
Cl, ppm (ar)	163	72
F, ppm (ar)	27	207
*ar as received		

Table 13 Analysis of palm kernel shells (PKS) and wood chips (Conti and others, 2008)

Proximate analysis	PKS	Wood chips
Moisture, wt% (ar*)	16.80	32.20
Ash, wt% (ar)	1.91	1.88
LHV, MJ/kg (ar)	15.7	13.2
Ultimate analysis		
C, wt% (ar)	49.39	51.84
N, wt% (ar)	0.26	0.15
H, wt% (ar)	5.71	6.2
O, wt% (ar)	42.32	39.01
S, wt% (ar)	0.02	0.03
Cl, ppm (ar)	319	52
F, ppm (ar)	11.84	na
* ar as received		

SCR for NO_x control. The plants fire a combination of Indonesian and South African coals and the characteristics of these coals are summarised in Table 12. Conti and others (2008) report on experiments firing two biomass materials (palm kernel shells – PKS, and pine chips) with the usual blend of Indonesian and South African coals. The characteristics of the biomass materials are summarised in

Table 13. The biomass was added at relatively low concentrations (1.1% PKS and 3.3% pine chips) based on the total plant energy input. The emissions for the plant were compared with the Italian legislation on emissions for large coal-fired plants (>500 MWe).

The energy efficiency of the plant when running on coal alone is around 38% and it was reported that the cofiring of the biomass fuels did not affect this significantly. Palm kernel shells (PKS) were fired at a constant rate of 1.4 t/h which was the maximum allowed by the existing mill. The PKS was mixed directly with the coal and fed into the boiler during three different but identical test runs. Modifications to the fuel feed system were required to allow the cofiring of the wood chips, which were also tested during three different runs. Tables 14 and 15 show the results for the different tests (cofiring in three trials with PKS then cofiring in three trials with wood chips) comparing the emissions from the plant with the limits defined in Italian law. The reference case in each study is the firing of the coal alone. It is not clear why emissions from the reference coal burn in Table 14 differ from those in Table 15 – no explanation was given by Conti and others (2008). But it may well have been the higher base coal emissions during this study that were responsible for some of the higher emission rates for some of the trace species rather than the wood chips. For example, the dioxin emissions during the co-combustion of wood chips was reported to be higher than that from the combustion of PKS but this was probably largely due to the fact that the coal fired during this study was producing a higher concentration of dioxin emissions. It would seem that the cofiring of the wood chips actually reduced the dioxin emissions in tests 2 and 3. Then again, the emission levels being measured throughout both studies for species such as dioxins were either so low and so close to, or below, the detection limit that these variations may well have been due to analytical variation and error rather than due to any actual change in the emissions. There were a few other discrepancies that were not explained. For example, it was not clear why the PM₁₀ emissions were so high in the third test run using wood chips. Presumably this was due to some temporary problem with the particulate control system.

All emissions for all the biomass co-combustion test runs were reported to be well below the limits required by Italian law. If anything, the study served to emphasise that the variability of trace elements in coal is significant and that emissions can be subject to changes in plant performance as well as changes in fuel chemistry.

4.3 Minnesota Power Rapids Energy Centre and Iowa Main Power Plants, USA

A previous IEA CCC report (Fernando, 2005) reviewed cofiring experience at several full-scale coal-fired plants in the USA, each of which showed little or no detrimental effects on emissions of the major pollutants considered (particles, SO₂ and NO_x). Although this report is aimed mainly at full-scale coal-fired plants, it would appear that most full-scale plants do not have significant issues with the emissions from co-combustion. By looking at emissions from smaller,

Table 14 Results of cofiring PKS at the Fiume Santo plant, Italy (Conti and others, 2008)

Trial	PM ₁₀	TSP	NH ₃	HCl	HF	HBr	TOC	NOx	SO ₂	CO	CH ₄	PAH	PCCD/ PCDF	
	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	µg/m ³	ngTEQ/m ³	
Ref (coal)	0.77	1.36	<0.20	1.92	<0.50	<0.50	<0.80	136	326	17	<1.00	<0.06	0.0001	
1	0.54	1.46	<0.20	1.38	<0.50	<0.50	<0.80	133	328	54	<1.00	<0.06	0.0001	
2	0.50	1.50	<0.20	1.25	<0.50	<0.50	<0.80	157	183	11	<1.00	<0.06	0.0003	
3	2.02	3.32	<0.20	1.53	<0.50	<0.50	<0.80	169	109	15	<1.00	<0.06	0.0001	
Italian Law	–	50	100	100	5	5	300	200	400	250	–	10	0.1	
Trial (all mg/m ³)	As	Cd	Co	Cr	Cu	Hg	Mn	Ni	Pb	Sb	Sn	Tl	V	Zn
Ref (coal)	<0.0010	<0.0001	<0.0006	<0.0006	<0.0003	<0.0029	<0.0006	<0.0006	<0.0006	<0.0006	<0.0013	<0.0006	<0.0006	<0.0017
1	<0.0010	<0.0001	<0.0005	<0.0013	<0.0002	<0.0022	<0.0005	<0.0011	<0.0005	<0.0005	<0.0010	<0.0005	<0.0005	<0.0027
2	<0.0010	<0.0001	<0.0005	<0.0020	<0.0002	<0.0023	<0.0012	<0.0015	<0.0005	<0.0005	<0.0010	<0.0005	<0.0005	<0.0039
3	<0.0010	<0.0001	<0.0005	<0.0007	<0.0003	<0.0022	<0.0011	<0.0006	<0.0005	<0.0005	<0.0009	<0.0006	<0.0005	<0.0033
Italian Law	1	0.2	1	1	10	0.2	10	1	10	10	10	0.2	10	–

Table 15 Results of cofiring wood chips at the Fiume Santo plant, Italy (Conti and others, 2008)

Trial	PM ₁₀	TSP	NH ₃	HCl	HF	HBr	TOC	NOx	SO ₂	CO	CH ₄	PAH	PCCD/ PCDF	
	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	mg/m ³	µg/m ³	ngTEQ/m ³	
Ref (coal)	7.80	12.1	<0.20	2.35	1.70	<0.50	1.44	133	283	16	<1.00	<0.06	0.0003	
1	5.97	11.2	<0.20	1.87	1.59	<0.50	1.27	161	317	12	<1.00	<0.06	0.0001	
2	4.82	10.1	<0.20	2.05	1.08	<0.50	1.27	159	320	9	<1.00	<0.06	0.0002	
3	535	10.8	<0.20	2.09	1.79	<0.50	1.83	141	313	12	<1.00	<0.06	0.0001	
Italian Law	–	50	100	100	5	5	300	200	400	250	–	10	0.1	
Trial (all mg/m ³)	As	Cd	Co	Cr	Cu	Hg	Mn	Ni	Pb	Sb	Sn	Tl	V	Zn
Ref (coal)	<0.0015	<0.0001	<0.0010	<0.0013	<0.0013	<0.0023	<0.0088	<0.0023	<0.0023	<0.0010	<0.0020	<0.0008	<0.0033	<0.0046
1	<0.0020	<0.0001	<0.0010	<0.0014	<0.0014	<0.0020	<0.0032	<0.0022	<0.0012	<0.0010	<0.0020	<0.0010	<0.0062	<0.0046
2	<0.0026	<0.0001	<0.0010	<0.0010	<0.0013	<0.0020	<0.0028	<0.0018	<0.0010	<0.0010	<0.0020	<0.0010	<0.0067	<0.0041
3	<0.0013	<0.0001	<0.0010	<0.0009	<0.0012	<0.0022	<0.0027	<0.0019	<0.0011	<0.0011	<0.0021	<0.0011	<0.0061	<0.0042
Italian Law	1	0.2	1	1	10	0.2	10	1	10	10	10	0.2	10	–

Table 16 Emissions from the US EPA ETV studies on biomass cofiring (SRI/USEPA, 2008a,b)

	Minnesota Power's REC boiler coal + wood waste (92:8 by weight)			University of Iowa Boiler 10 coal + wood pellets (85:15 by weight)		
	baseline average	cofire average	difference	baseline average	cofire average	difference
Heat input, MW	88	106	21.8%	78	80	3.00%
Heat output, MW	65	65	0.00%	66	67	2.10%
Efficiency, %	74.5 ± 0.005	61.3 ± 0.7	-17.7%	84.9 ± 0.4	84.1 ± 0.7	-0.09%
Total particulates	0.0045 ± 0.0004	0.0060 ± 0.003	-81.2%	0.0061 ± 0.03	0.044 ± 0.003	-28.1%
Filterable particulates	0.0044 ± 0.0004	0.0037 ± 0.002	-17.1%	0.031 ± 0.008	0.024 ± 0.0018	-22.8%
Condensable particles	0.0249 ± 0.0013	0.0034 ± 0.0015	-86.5%	0.030 ± 0.02	0.020 ± 0.0012	-33.9%
CO ₂	160 ± 7	131 ± 4	-18.3%	205 ± 2	207 ± 0.3	0.82%
SO ₂	0.474 ± 0.02	0.0013 ± 0.0001	-99.7%	2.47 ± 0.14	2.16 ± 0.08	-12.4%
NO _x	0.527 ± 0.01	0.194 ± 0.007	-63.2%	0.460 ± 0.02	0.506 ± 0.018	10.2%
CO	0.230 ± 0.02	0.555 ± 0.2	142%	0.088 ± 0.010	0.083 ± 0.05	-5.02%

industrial-scale boilers, with lower combustion efficiencies, we are potentially more likely to see significant changes in plant performance and emissions.

In 2006 the US EPA's Environmental Technology Verification Program (ETV) initiated a study to determine the feasibility of biomass cofiring in coal-fired boilers. The programme concentrated on two industrial-commercial-institutional (ICI) boilers:

- Minnesota Power's Rapid Energy Centre (REC) boiler cofiring bark with coal;
- University of Iowa (UoI) main power plant boiler cofiring wood derived pelletised fuel with coal.

The REC facility has two spreader stoker boilers firing western subbituminous coal each at around 175,000 lb/h (just under 80 t/h, 15 MWe). The plant provided energy to the neighbouring paper mill. Waste wood and bark from the mill and from other local facilities was cofired with the coal at a ratio of around 8:92 by weight. The comparison of the baseline emissions with those from cofiring are summarised in Table 16. The SO₂ emissions were reduced significantly (over 99%) and NO_x emissions were also reduced by 63.2%. The emission of particles also dropped by over 17% and the condensable material by over 85%. Condensable particulate matter is material which appears in the gaseous form on emission but which rapidly condenses to form particles in the plume from the plant (SRI/US EPA, 2008a).

Although not included in the table, the emissions of metals were also reported to be reduced with the reductions in Hg and Se reported to be 'statistically significant'. Reductions in HCl and HF were 62% and 72% respectively. The only detrimental effect from the cofiring of the waste wood material was in the fly ash where the changes in ash characteristics were reported as significant. Although the leachability of the fly ash did not exceed the legislated limits, the ash itself did not meet the US EPA Class F requirements

for use in concrete. However, the ash from coal combustion alone at the plant had not met these requirements either (SRI/US EPA, 2008a).

Pelletised wood fuel (pressed oak product) provided by Renewafuels LLC was tested at the UoI facility. The UoI Power plant is a combined heat and power facility producing energy from four boilers – one stoker unit (Boiler 10), one CFBC and two gas boilers. Boiler 10 was used for this study. Boiler 10 produces around 206 MMBtu/h (60 kW/h) and is fitted with an ESP. The results from cofiring coal with the pelletised wood fuel were included in Table 16. The reduction in emissions of particulates, SO₂ and NO_x was less dramatic than that seen at the REC. As with the REC study, neither the ash produced from coal combustion alone nor with biomass combustion were suitable for use in cement or concrete applications.

Both the REC and UoI studies were aimed at testing the feasibility of biomass co-combustion and the potential effect on CO₂ emissions. The CO₂ reduction for the ROC plant was calculated to be 90%. The reduction at the significantly smaller UoI plant was estimated at 10%. The reductions in the other pollutants listed in Table 16, such as SO₂ and NO_x, are, once again, a fortunate co-benefit of cofiring biomass with coal.

4.4 Comments

Drax is currently the largest pulverised coal fired plant in the world which is cofiring biomass with coal. The operation of the plant has been under close scrutiny during the test periods and is still subject to tightened requirements for emissions monitoring during its continued operation. Various biomass materials have been sourced from around the world with different characteristics. However, emissions of particulates, SO₂ and NO_x have been consistently lower with biomass

cofiring. Emissions of trace elements to both air and other media (water and ash) have also been reduced with biomass co-combustion at the plant.

Two units at the Fiume Santo plant in Italy are cofiring biomass such as palm kernel shells and pine chips with coal. Emissions of pollutants, although variable, are consistently below those limits set in the Italian legislation.

Studies at two industrial coal-fired units in the USA demonstrated that, even in coal combustion systems with lower efficiency than full-scale pulverised coal plants, the addition of biomass as a fuel resulted in significantly lower emissions of particulates, SO₂ and NO_x. Although the co-combustion did result in negative effects on the fly ash, the fly ash was never saleable anyway.

5 Legislation and incentives

Much of the impetus behind the cofiring of biomass with coal is due to some form of legislative requirement or incentive. In many cases, the problems associated with co-combusting biomass and waste, such as handling problems, sourcing, slagging and fouling and so on, would mean that plant operators would simply not consider making the necessary changes. Plants are being pushed towards co-combustion options with legislation and/or incentives. Sections 5.1 and 5.2 briefly review some of the incentives in place in different countries. The Sections also include, where possible, details of other legislation (such as emission limits and restrictions on fly ash use) which would impact on the decision at a plant on whether to cofire biomass or waste with coal.

5.1 EU

The EU has set a minimum requirement for renewable energy (20% energy consumption in 2020 to come from renewables). This would mean around 195 Mtoe of biomass as compared with the current (2005) total of 85 Mtoe, contributing around 5% of the energy consumption in the EU-27 (current 27 members of the EU). In both cases, biomass accounts for around two-thirds of the renewable energy total (Kautto and Peck, 2008). However, there seems to be, as yet, no single policy on biomass promotion within the EU. Some countries within the EU, such as the Netherlands and the UK, actively promote biomass use at coal-fired plants whereas others, such as Germany, do not. Ireland has its own biomass action plan including a target of 30% cofiring of biomass in peat stations by 2015. It seems that there is a case against subsidising biomass cofiring with the argument being that this would also lead to increased profitability of existing coal-fired plants which would extend the use of fossil fuels. It is estimated that between 50 and 90 TWh/y is produced from biomass use in electricity production in the EU27 (Lintunen and Kangas, 2009).

Feed-in laws are the most common policies for promoting renewables in the EU and have been implemented in 21 EU member states. There are two main types of feed-in laws (Lintunen and Kangas, 2009):

- feed-in tariffs (FIT) set a minimum price for electricity produced from renewables;
- feed-in premiums (FIP) are an extra price on top of the electricity price for renewable energy production.

Six EU countries also have tradeable green certificates such as the ROC (Renewable Obligation Certificate) scheme in the UK. Sweden has an energy certificate scheme in which suppliers are obliged to acquire electricity certificates from renewable plants equal to the percentage of electricity they supply. Denmark has a tradeable renewable energy certificate scheme. This means that, although using different policies, all EU member states have some form of financial incentive to promote renewable energy production. In most cases, the increased cost is passed on to the electricity consumers (Fernando, 2005).

Lintunen and Kangas (2009) have modelled the effect of different feed-in laws and found that, as expected, financial incentives to use biomass promote biomass cofiring. However, this increase in biomass use does not necessarily lead to increased fossil fuel use and therefore these schemes would not necessarily work against climate policy goals, as suggested above. In fact, cofiring of biomass with coal offers a cost-efficient short-term solution for increasing the share of renewable electricity in the EU while alternative carbon neutral power generation technologies are being developed.

Kangas and others (2009) used mathematical models to evaluate the effect of feed-in tariffs on the use of biomass (wood) at two hypothetical power plants – a pulverised coal fired plant in a Central European country versus a co-generating peat-fired FBC plant in a Nordic country. The study showed that, without promotional policies, it is not profitable to cofire wood and coal in the pulverised coal fired plant. There is, however, a threshold (varying with power demand) for the price of CO₂ emission credits beyond which wood becomes optimal. The feed-in tariff would appear to be less important in the economics of the pulverised coal fired plant than the CO₂ credit price. For the Nordic FBC scenario, due to factors such as the availability of the fuel, it would always be profitable to cofire wood with peat.

The use of biomass in cofiring is not normally economic in large-scale plants since biomass fuels tend to be more expensive. Some countries, such as the Nordic countries, do find it profitable to combust wood, forest and logging industry wastes with peat, when the sources are close to a power plant. According to Lintunen and Kangas (2009) biomass based electricity will be the lead form of renewable energy in the Nordic countries during the next decades. The move away from fossil fuels to comply with low-carbon policies will increase the demand for biomass materials such as wood and make the use of biomass even more expensive. Policies which promote the use of biomass are therefore necessary to promote cofiring (Kangas and others, 2009).

The tightening of EU directives on waste disposal will mean that sources will be searching for alternative means to dispose of waste materials. For example, the change in EU legislation on landfill means that this method of disposal is no longer an option for sewage sludge. One of the possible routes for sewage sludge treatment is valorisation through co-combustion with coal (Barbosa and others, 2009). Waste tyres can also no longer be sent to landfill in the EU and the cofiring of waste tyres with coal is increasing (Singh and others, 2009).

The EU has stringent emission limits for all combustion plants and incinerators. These are summarised in Table 17. Over and above this, individual plants are subject to site-specific permits which may mean even more stringent control requirements. Emission legislation was discussed in more detail in a previous report from IEA CCC (Sloss, 2009).

The EU's Large Combustion Plant Directive (LCPD) establishes emission limits for major pollutants according to whether the plant is existing or new. Newer plants must meet more stringent limits than older plants. The LCPD does not cover cofiring of waste fuels that are covered by the EU's Waste Incineration Directive (WID; *see below*) but does cover biomass fuels such as vegetable waste from agriculture, forestry and food processing, fibrous waste from the paper and pulp industry, cork waste, and wood waste (other than

those with protective treatments or coatings containing more hazardous halogenated or organic compounds). If these materials are cofired then the original LCPD limits still apply.

If co-combustion of other waste is to be applied at an existing coal-fired plant then the emission limits must be revised according to the EU Waste Incineration Directive WID (2000/76/EC). Table 18 includes a summary of the EU emission limits under the WID alongside the emission limits in Germany (*see* Section 5.1.2 below). The EU WID limits are more stringent than those in the LCPD. The 'mixing rule' is required to determine the emission limits of organic compounds, HCl and HF when biomass is cofired at large (>300 MWth) coal-fired plants. Under the mixing rule, plants firing waste materials defined under the WID must calculate a specific emission factor based on the amount of waste material being cofired. The emission limits are then a weighted value of the limits for 100% of each fuel. The weighting factor is the flue gas volume produced during mono-combustion of either of the fuels. The emission limit (EL) for each pollutant (I) is then calculated using the following equation (Leckner, 2007):

$$EL_{\text{imix}} = (V_w EL_{\text{iw}} + V_{\text{bf}} EL_{\text{ibf}}) / (V_w + V_{\text{bf}})$$

	Combustion plants		
	Solid fuels	Biomass	Waste incineration
CO, mg/m ³	local directives		50
NO _x , mg NO ₂ /m ³	300	300	200
SO _x , mg SO ₂ /m ³	525	200	50
Hg, mg/m ³	0.05	0.05	0.05
Ref O ₂ vol, %	6	6	11

Compound	13th BlmSchv*	17th BlmSchv*		EU-Directive			
	Power plants	Waste incineration		Waste incineration		Co-combustion	
		Daily average (6% O ₂)	Daily average (11% O ₂)	Half-hour average (11% O ₂)	Daily average (11% O ₂)	Half-hour average (11% O ₂)	Power plants >300 MWth
CO, mg/m ³	250	50	100	50	100	mixing rule	–
Organic compounds as total C, mg/m ³		10	20	10	20	mixing rule	10
Particulate matter, mg/m ³	50	10	30	10	30	30	30
SO ₂ /SO ₃ as SO ₂ , mg/m ³	400	50	200	50	200	200	50
Sulphur emission factor, %	15					5	
NO _x , mg/m ³	800	200	400	200	400	200	800
HCl, mg/m ³	100	10	60	10	60	mixing rule	10
HF, mg/m ³	15	1	4	1	4	mixing rule	1
Σ TI +Cd, mg/m ³		0.05		0.05		0.05	0.05
Hg, µg/m ³		0.03	0.05	0.05		0.05	0.05
Σ Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V, Sn, mg/m ³		0.5		0.5		0.5	0.5
PCDD/PCDF, ng/m ³		0.1		0.1		0.1	0.1

* Bundesimmissionsschutz-Verordnung or BlmSchv limits

Where:

V_w exhaust gas volume from waste only at 11% oxygen content, m³/h

V_{bf} exhaust gas volume from the base fuel (coal) only at 6% oxygen content, m³/h

EL_{iw} emission limit for pollutant i in a waste combustion plant, mg/m³

EL_{ibf} emission limit for pollutant i for power plants given in the EU Directive, mg/m³

Examples of the application of this Directive are given in Figure 17. The measured values for emissions from cofiring wood with coal and wood with sewage sludge in a CFBC boiler are compared with the emission limits set by the EU

WID Directive. It is clear from Figure 17a that co-combustion of wood with sewage sludge cannot meet the WID limits for SO₂ when cofiring more than around 10% sludge and will not meet the NO limits when firing more than 30% sludge. However, cofiring sewage sludge with coal is less likely to cause an infringement of the WID limits as the SO₂ limit would only be exceeded at over 45% sewage sludge and the NO limit at around 35% sewage sludge, as shown in Figure 17b. These results were from an unnamed CFBC boiler in Sweden. This study demonstrates potential issues for cofiring sewage sludge in CFBC systems. However, since we know (from Chapters 2 and 3) that fuel characteristics and combustion conditions can have a significant effect on emissions, it is likely that different plants may give different results. Monitoring of emissions is therefore recommended on plants switching to co-combustion to ensure that emission limits are not exceeded.

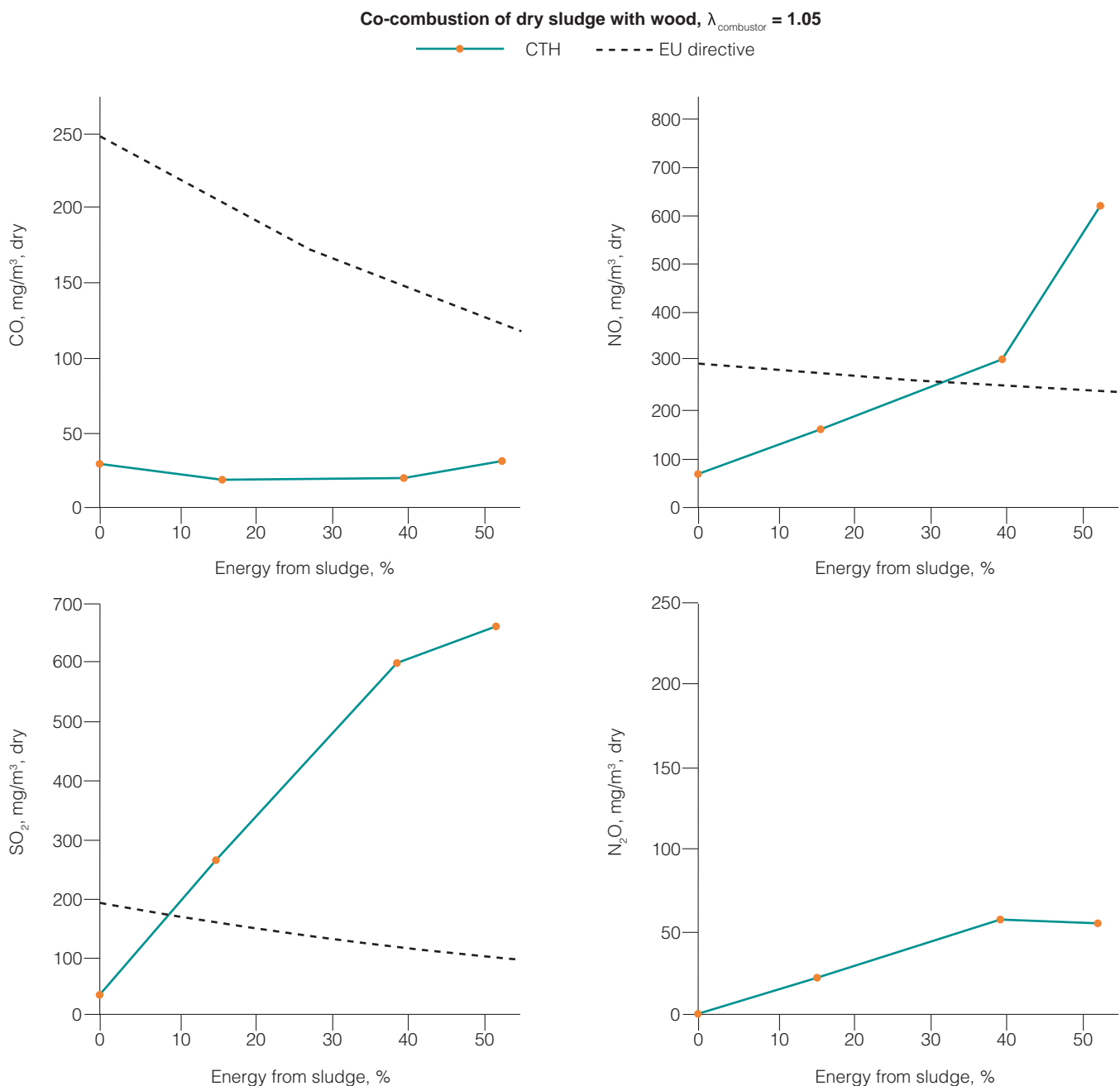


Figure 17a Comparison between measured values in a CFB boiler firing wood and sewage sludge and the EU Waste Incineration Directive values (Leckner, 2007)

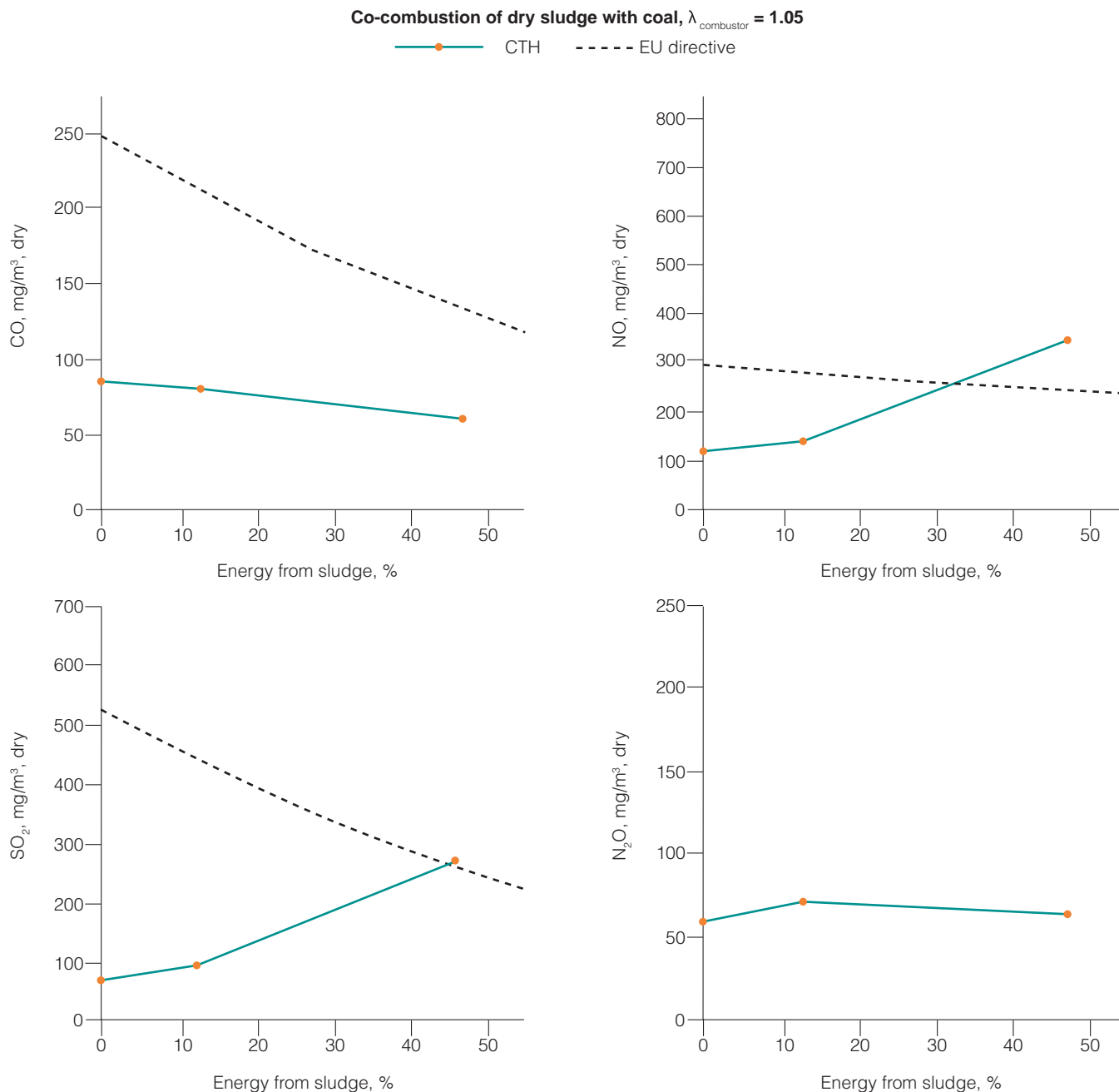


Figure 17b Comparison between measured values in a CFB boiler firing coal and sewage sludge and the EU Waste Incineration Directive values (Leckner, 2007)

In addition to the more stringent emission limits, the WID also has more stringent monitoring requirements, including continuous monitoring for the major species (SO₂, NO_x, particulates, CO, TOC (total organic carbon), HCl, HF and flue gas moisture). The WID also has more rigorous requirements relevant to plant operation than the LCPD, such as minimum combustion temperatures (Fernando, 2007). The WID is far more challenging with respect to compliance than the LCPD and so existing coal-fired plants are generally reluctant to cofire waste material that falls under the remit of the WID.

In the study by Hansson and others (2008) on the potential for biomass cofiring with coal in the EU, the issue of ash was mentioned. It was recognised that the existing standards in Europe (such as ENV 450) do not allow the use of ash produced from the cofiring of biomass with coal. This was

discussed in Section 3.1.8. Since many plants make money from the sale of fly ash to cement and concrete production, the potential loss of income could be a significant factor in determining whether to switch to co-combustion. Denmark has already relaxed the current law to allow the use of some biomass fly ash in concrete production (Fernando, 2005). However, there is a new European standard being developed to replace the old standard. The new standard would restrict the allowable cofiring ratio and would demand that the suppliers of the ash demonstrate that the ash from cofiring behaves similarly to coal ash.

The following sections review relevant legislation in example countries to show the types of challenges being faced by plant operators who choose to cofire biomass and waste with coal.

5.1.1 Bosnia

Bosnia has lignite and brown coal reserves which are used at the four large coal-fired power plants in the country. These fuels are low ranking, variable, low in calorific value, high in sulphur and prone to causing slagging and fouling at the plants. Wooden biomass material is available in the region from forests and local sawmills. Kazagic and Smajevic (2009) studied the effects of cofiring local wood-based biomass materials with the different Bosnian coals and lignites in a laboratory-scale pulverised fuel entrained flow reactor at Sarajevo University. It was found that cofiring wood biomass with the Bosnian coal could reduce SO₂ emissions (up to 28%). However, the cofiring of sawdust with lignite did not have this effect. Although the nitrogen content of the fuels differed, as discussed in Section 3.1.3, it appears to be the combustion conditions which are more important in determining final NO_x emissions. Further work was called for to identify the most appropriate fuel blends and combustion conditions to maximise the potential benefits of cofiring biomass in Bosnia.

5.1.2 Germany

Germany commonly has national standards which are more stringent than those set at the EU level. In addition to the EU LCPD and WID, Germany has its own Ordinance on Large Power Plants and an Ordinance on Incineration for Waste and Similar Combustible Materials. As with the LCPD and WID, the German standards also allow for a calculation of emission limits based on the proportion of biomass used. However, the German mixing rule also takes into account the calorific values of the fuels as well as the flue gas volumes and other data. In addition to this, extensive testing and measurement may be required at each plant to obtain a permit to allow cofiring. A summary of selected emission limits were included in Table 18 alongside the emission limits in the EU, as discussed above.

Germany has around 20 plants cofiring sewage sludge with coal, as summarised by Fernando (2007). This is largely due to the tight legislation covering landfill in Germany which makes cofiring sewage sludge with coal more appropriate than elsewhere.

5.1.3 Hungary

Hungary set a target of up to 3.6% of gross total electricity to be 'green' by 2010 and, by 2008, had surpassed this target with 354 MW of installed capacity cofiring biomass. The cofire of biomass material at many of the existing plants meant that these plants could continue to operate despite previous plans to close them down. Although this move to cofire wood with coal at the plants is successful with respect to achieving green targets and reducing CO₂, there are problems due to the disturbance of the balance of the wood market in the region and due to disagreements over the specification of wood as a green energy. Grabner and Toth (2008) suggest that, despite recent changes in Hungarian

regulations on biomass and co-generation, more work is needed to harmonise the activities of the different regulatory acts and bodies to ensure that the use of biomass in Hungary is not hindered in future.

5.1.4 The Netherlands

The Netherlands have a similar mixing rule to that in Germany for cofiring waste in large coal-fired systems which is more stringent than that in the EU WID. Further, the limits for groups of the heavy metals Cd, Tl, Hg and the sum of a group of nine others (As, Co, Cr, Cu, Mn, Ni, Pb, Sb and V) are stricter than those in the EU WID (Fernando, 2007).

The Dutch Ministry of Economic Affairs have established a long-term energy research strategy programme which includes ambitious targets to achieve 25% and 40% biomass cofiring levels in 2020 and 2040 respectively along with coal+biomass-to-energy efficiencies of over 50%. Kiel and others (2009) emphasise that achieving this will be a major challenge which will involve the development of new and improved technologies. Further, in order to maintain the country's admirable rate of 100% use of coal ash, research and development is necessary to ensure that ashes from cofiring can be used. Biomass ashes are physically and chemically different from coal ashes.

According to de Wilde and others (2007) emissions of PM₁₀ from industrial biomass-based sources will have increased by a factor of 3 from 140 t/y to 470 t/y between 2004 and 2020. Although some biomass combustion will be in dedicated biomass combustors, the dominant sources of the increased emissions will be the co-combustion of biomass in coal plants and bio-oil in large diesel plants. However, even with this increase the contribution to total emissions of PM₁₀ from human activities in the country will still be around only 1%. Although the substitution of biomass for coal in cofiring systems will likely result in a reduction in actual PM₁₀ emissions from existing plants. The overall increase in PM₁₀ emissions will be as a result of the overall increase in coal and biomass use and not due to the substitution of coal with biomass. However, there is likely to be a change in the size-distribution and chemical composition of the particles emitted with a possible increase in PM_{1.0}. de Wilde and others (2007) emphasise that little is known on this issue and more research is needed.

5.1.5 Spain

The Spanish National Renewables Energy Plan aims to produce 12% of the energy generation of Spain in 2010 from renewable energies and almost half of this will be from biomass. In the study by Gasol and others (2008) on the environmental impact of this Energy Plan, the focus was on the effect of land use, irrigation and fertiliser use, with emission changes not being mentioned. Royo and others (2008) summarised the challenges to be faced in Spain with respect to implementation of the Energy Plan and the studies being performed to assess the potential for cofiring in Spain, based on life cycle analysis. Changes in emissions of SO₂ and

NO_x were briefly mentioned in the paper but were clearly not considered a particular barrier to cofiring. That is not to say that these studies do not believe that emissions will change as a result of cofiring, rather that they do not see the potential changes as a problem that would hinder the further expansion of cofiring in the country.

5.1.6 UK

The UK has had, since 2002, a Renewables Obligation scheme for expanding the use of renewable energy. Under the scheme coal-fired plants receive one Renewables Obligation Certificate (ROC) for each megawatt hour of electricity they generate by cofiring biomass. The ROC allowance continues to apply to purpose-grown energy crops but the ROC value is different for other biomass materials (Marshall, 2009). In order to limit the effect of large-scale cofiring on the ROC market, restrictions are placed on the fuels used at cofiring stations as well as the amount of cofiring that is allowed. From April 2006 until March 2011 only a maximum of 10% of the renewables obligation of any electricity supplier can be from the production of cofired ROCs. Between March 2011 and March 2016, this maximum will be lowered to 5%. This means that an increasing proportion of the biomass used must come from energy crops (Fernando, 2007).

The Drax plant in North Yorkshire (*see* Chapter 4) aims to use 12.5% biomass for cofiring from 2010 onwards. The EDF (Electricité de France) Cottam and West Burton plants in Nottinghamshire currently fire 0.8% and 1.2% biomass respectively. Only one of E.ON's three UK plants – Kingsnorth – is currently cofiring with 1.49% biomass (Marshall, 2009). Fiddler's Ferry has two 500 MWe boilers which have been designed to fire at up to 20% (thermal input) biomass (Boneham, 2008).

The challenge for UK suppliers seems to be the security of supply of sufficient quantities of biomass. In 2007, less than 5% of the biomass fired at Drax came from the UK. Drax does not reveal the source of its biomass as it does not want to alert competitors. Although it would be feasible to consider a 4 GW biomass-only plant at Drax, the storage of 15 Mt of biomass per year would require a substantial commitment to land procurement and development (Marshall, 2009).

Recent news coverage (such as BN, 2010) would suggest that there is an issue with the limitation of subsidies for biomass use in the UK. Whilst some wind-farms have been guaranteed 20 years of government support, the biomass plants at Drax have only been guaranteed support for four years. It is likely that the final decision on the Drax plants will be made later, subject to discussions with the Government.

Although cofiring 50,000 t/y of thermally dried sewage sludge with coal has been successfully tried at full scale at the 4 x 576 MWe Longannet Plant in Scotland, a change in legislation stopped this practice in 2004. A Judicial Review ruled that sewage sludge was a waste and that the plant would henceforth have to meet the limits of the EU WID. As a result of this, the application of the tighter WID emission limits made cofiring impractical. Following this result, it is unlikely

that any other UK plants would consider cofiring sewage sludge without either a change in policy or some other financial or economic incentive to do so (Fernando, 2007).

The co-combustion of RDF in CFBC systems was carried out in two boilers in Slough during the 1980s at 40 RDF:60 coal on a heat basis. However, this has ceased since ROCs can only be gained if the levels of contamination in the waste are precisely known. The RDF is now fired separately in a dedicated RDF combustion system (Fernando, 2007).

The cofiring of waste tyres has been tested at a pilot plant in the UK and demonstrated to be a clean source of energy, with beneficial reductions in NO_x emissions. However, the current classification of tyres in the Environment Agency's Waste Quality Protocol mean that there are problems with cofiring tyres with coal on full-scale plants. Singh and others (2009) argue that reclassification of tyres in the protocol could allow them to provide a renewable source of energy.

Scotland has an ambitious target of 18% electricity from renewable sources by 2010 rising to 40% by 2020 but has set no targets specifically for biomass (Kautto and Peck, 2008).

5.2 USA

There have been over 40 commercial cofiring demonstration plants in the USA. The evolution of tax credits for biomass use has been somewhat complex in the past. Regulatory and tax incentives have been used to promote biomass use in the USA since the 1990s. However, the only type of cofiring of biomass with coal that receives a federal tax credit is that for systems using 'closed-loop' biomass. Closed-loop biomass is defined as crops grown in a sustainable manner for the purpose of bioenergy and bioproduct uses. Open-loop biomass is defined as biomass not produced for such purposes, such as agricultural wastes and residues. Electricity produced from cofiring an open-loop biomass material with coal will only qualify for a tax credit if the amount of fossil fuel used is the minimum required for start-up and flame stabilisation – that is, where the fossil fuel is being used to establish combustion but not to maintain it. Therefore any coal plant cofiring open-loop biomass will not qualify for tax credits (Fernando, 2007).

At the moment it would appear that the greatest push towards biomass cofiring is within the state-by-state regulatory regimes. The wide-variety of plant designs, available fuel stocks and, in many cases, quite restrictive state regulations, will determine which plants will find it economic to move towards biomass co-combustion (Eisenstat and others, 2009).

There do not seem to be any mixing rules, similar to those seen in the EU, to determine emission limits for plants which switch to cofiring. The legislation in the USA tends to be set on a plant-by-plant basis with each plant required to meet individually determined limits set within a permit or authorisation. It is likely that the authorities take potentially detrimental changes in emissions due to cofiring into account before any permit is granted.

There is some national legislation which may affect plants switching to cofiring. As discussed in Section 3.3, the existing standard for fly ash use (ASTM C618) does not allow fly ash from anything other than coal combustion alone to be used in cement production. This means that some plants will lose revenue from the loss of ash sales and have a new solid waste stream to deal with should they choose to convert to cofiring.

It is estimated that 120 million tons (109 Mt) of biomass will be needed annually in the USA as a feedstock for advanced biofuels and bioenergy. It is suggested by Hinchee and others (2009) that biotechnology applications such as genetic modification and selective pollination may be required to ensure the supply of biomass is sufficient.

5.3 Comments

The cofiring of biomass at pulverised coal fired plants is being encouraged in several EU countries as part of the move towards greener energy. However, there is no single EU policy on co-combustion and so individual member states are making their own legislation and action plans, most of which are based on financial incentives such as feed-in-tariffs and premiums.

The EU sets standards for emission limits from all coal-fired plants in the EU under the LCPD. The EU has more stringent limits for waste incinerators under the WID. For plants cofiring materials specified as waste under the WID in full-scale plants, the emission limits change to a calculated combination of both the LCPD and WID under the mixing rule. In many cases, the resulting limits are regarded as very stringent and can be the deciding factor on whether a plant is prepared to consider co-combustion. In most cases, plants cofiring waste and biomass with coal at relatively low concentrations (such as below 20% or 30% biomass by weight) the standards are easy to meet. However, this is not the case when firing higher concentrations of more challenging wastes such as sewage sludge or RDF/MSW.

Several papers have been reviewed which suggest that current legislation relevant to cofiring is too stringent and is hindering the use of otherwise valuable co-combustion fuels. For example sewage sludge cofiring with coal was tested in Scotland but then abandoned due to tightening of the applicable emission limits in the UK. However, Germany cofires sewage sludge at around 20 plants because the tighter German legislation forbidding the dumping of sewage sludge to landfill makes this economically sensible. The current UK legislation does not allow the co-combustion of tyres with coal at full-scale plants when it has been demonstrated that, at restricted concentrations, these can be regarded as a significant renewable source of energy.

Previous legislation in the EU and the USA did not allow the use of fly ash from a plant cofiring waste or biomass with coal. This has been proven in several tests to be overly restrictive and that, although the fly ash from co-combustion systems may require slight changes in the volume of additives used in the preparation of cement and concrete, the exclusion of fly ash from cofired systems is resulting in the waste of a

potentially valuable resource. Denmark has already reduced the stringency of the legislation on the use of fly ash from co-combustion and the EU is currently reconsidering the requirements set out in EN450.

6 Conclusions

Biomass combustion alone is seen as a potentially polluting activity since it is commonly burned in a relatively inefficient manner and can lead to significant emissions of some pollutants, especially organic material and particulate matter. This leads to concern that the cofiring of biomass with coal will result in greater emissions from the coal facility as a result. However, it would appear that this is not the case and that the higher combustion efficiency of coal combustion systems promotes the cleaner combustion of biomass and result in reduced pollutant emissions in most situations.

In the papers reviewed for this report, any listing of the advantages and disadvantages of cofiring biomass with coal tended to concentrate on cost, efficiency, and practical issues such as plant modification for fuel feeding systems and negative effects due to slagging and fouling. The issue of negative effects on emissions was rarely mentioned in discussion papers and only appeared in more specialist scientific papers. This would imply that the issue of changes in emissions due to cofiring is not considered a significant problem. In fact it is generally held that cofiring biomass with coal in comparison with firing coal alone actually leads to consistently lower particulate, SO₂ and NO_x emissions and emissions of trace species. In the studies reviewed, the measured emissions did not exceed any current international or national emission standards or legislation. There are two statements which may be made with respect to co-combustion:

- cofiring biomass with coal can be considered as a valid technique to reduce NO_x emissions;
- the co-combustion of biomass with coal can be regarded as a method of harnessing the energy present in the biomass whilst reducing emissions that would have been far more significant had the biomass material been burned alone.

The two statements listed above appear to hold true for all biomass materials such as wood, energy crops and agricultural wastes. However, more complex materials such as sewage sludge and RDF or MSW can prove to be more of a challenge. Higher concentrations of sulphur, halogens and trace elements in sewage sludge and waste derived fuels can cause plant issues, although these issues are again associated more with plant performance than emissions. However, studies have suggested that cofiring sewage sludge at greater than 30–40 wt% in CFBC systems could result in emissions of SO₂ and NO_x which exceed EU limits. This is unlikely to be the case for pulverised coal fired plants firing, as they tend to do, below 20% waste material. The higher concentrations of halogens and trace elements in sewage sludge and waste materials tend not to result in significantly higher emissions of these species in the flue gases except at high sewage sludge:coal ratios. It is likely that increased emissions at elevated sewage sludge mixing ratios could be dealt with by existing or retrofitted control technologies such as FGD systems. Trace element concentrations in sewage sludge co-combustion can also result in higher concentrations of these species in the waste streams, especially the fly ashes.

The elevated concentrations of trace elements in fly ash do not currently cause any negative effects on cement or concrete produced, although higher quantities of aerating chemicals may be required. However, previous standards in the EU specified a complete ban on the use of fly ash produced from co-combustion. Many researchers have argued that this restriction excludes the use of an otherwise valid resource. Denmark has already made changes to reduce this restriction and the EU is currently working to relax the EN450 standard.

The practice of cofiring sewage sludge in individual countries appears to be far more dependent on the national definition of such biomass material and on applicable landfill legislation than on any negative effect on emissions. For example, landfill legislation in Germany dictates that the cofiring of sewage sludge is the most appropriate disposal method for sewage sludge and, as a result, around 20 coal plants cofire the sludge. Conversely, in the UK sewage sludge has been defined as a waste and this invokes the limits of the EU WID which makes it difficult for plants cofiring sewage sludge to meet the tightened emission limits. Similarly, the cofiring of RDF with coal in the UK has ceased due to the tightening of legislation relating to the characterisation of ROC materials. It can be argued that the current generic legislation on the use of some of these materials (such as sewage sludge and tyres) and their strict definitions as ‘waste’ is negating what could otherwise be a relatively clean source of energy.

It can be concluded that the cofiring of most biomass and waste materials with coal in pulverised coal fired plants results in reduced emissions of major pollutants and emission levels of halogens and trace elements which do not exceed or even approach current emission limits. These materials tend to be concentrated into the ash and other waste streams. Although biomass and waste materials are known to be highly variable in their chemistry, it would seem that the majority of these materials can be a valuable source of clean energy to replace coal in full-scale combustion systems. Fuels such as these could be tested on a case-by-case basis to determine whether they can in fact be used as a carbon-neutral fuel without any detrimental effects on emissions and ash sales.

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