

Environmental impact of energy recovery from waste tyres

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Abstract: Accumulation of millions of worn automotive tyres poses a considerable environmental problem. As an important part of the solid waste stream in today's society, worn tyres have traditionally been discarded in landfills or stored in stockpiles. Over the past several decades, however, innovative alternatives to disposal have been developed, partly as a result of high tipping fees charged by landfill operators. Because of their high heat content and their low levels of moisture and nitrogen compared to coal, tyres are ideally suited for energy recovery through combustion. Utilization of waste tyres as supplemental or alternate fuel in various industrial combustion facilities, thus, has become one of the most important alternatives to disposal. Combustion processes, however, generate gaseous pollutants and solid waste materials, which must be disposed of or re-used as secondary raw materials. It is therefore important to characterize these combustion products in order to assess the environmental impacts of energy recovery from scrap tyres. Studies have shown that substantial reductions of some environmental pollutants can be achieved by partially replacing conventional fuels with waste tyres. On the other hand, using tyres as fuel may lead to considerable increases in the levels of other pollutants. Most notable among the effects of tyre combustion are, relative to conventional fuels, a pronounced decrease in the emission of nitrogen oxides into the atmosphere, and a generally significant increase in atmospheric zinc emissions as well as in the zinc contents of the solid combustion products. The geochemical effects on solid and gaseous combustion products are more or less pronounced depending on fuel composition, conditions of combustion, type of facility, and effectiveness of air pollution control devices. Thus, the use of tyre fuel has environmental impacts that must be weighed against the benefits of reducing the large volume of waste tyres in the global waste stream.

Every year, millions of worn tyres are discarded worldwide. In 2001 alone, Americans disposed of 281 million car, truck, bus, and aeroplane tyres, equivalent to approximately one discarded tyre per person (RMA 2002). The large amounts of tyres removed from vehicles each year make proper waste management vital. Traditionally, discarded tyres have been treated as waste and placed in landfills or stockpiles; however, in many cases they are also illegally dumped. In 1990, approximately 65.5% of the 278 million tyres discarded in the USA were disposed of in landfills or added to existing stockpiles; only 34.5% were reused, recycled or recovered for various commercial and industrial applications (Blumenthal 1993). Since then, it has been recognized that old tyres represent a resource rather than a waste material, and thus, various solutions for re-using or recycling tyres have been, and are currently being, developed (Fig. 1). As a result, the number of tyres being disposed of in landfills has decreased dramatically over the past decade, reaching 10% in 2001 (Table 1).

One of the main reasons for this decrease is that scrap tyres are an important source of energy. In the USA, for example, energy recovery from tyres increased from 11% of the total number of scrap tyres in 1990 to approximately 40% in 1994, and still remains at approximately this level now (Fig. 2). Today, energy recovery through combustion is the single most important market for discarded tyres in the USA (Table 1, Fig. 3). This procedure is also extensively applied in many other countries, including Japan and Korea (Jang *et al.* 1998; JATMA 2002), Canada (Polasek & Jervis 1994), and several European nations (Giugliano *et al.* 1999; BUWAL 2001; Mukherjee *et al.* 2003; NARRA 2003). It is therefore important to assess the environmental impacts of energy recovery from scrap tyres.

This chapter will present a short overview of worn tyre management practices, with data for the USA, followed by a discussion of the effects of energy recovery through tyre combustion on both solid waste products and atmospheric emissions. In addition to reviewing

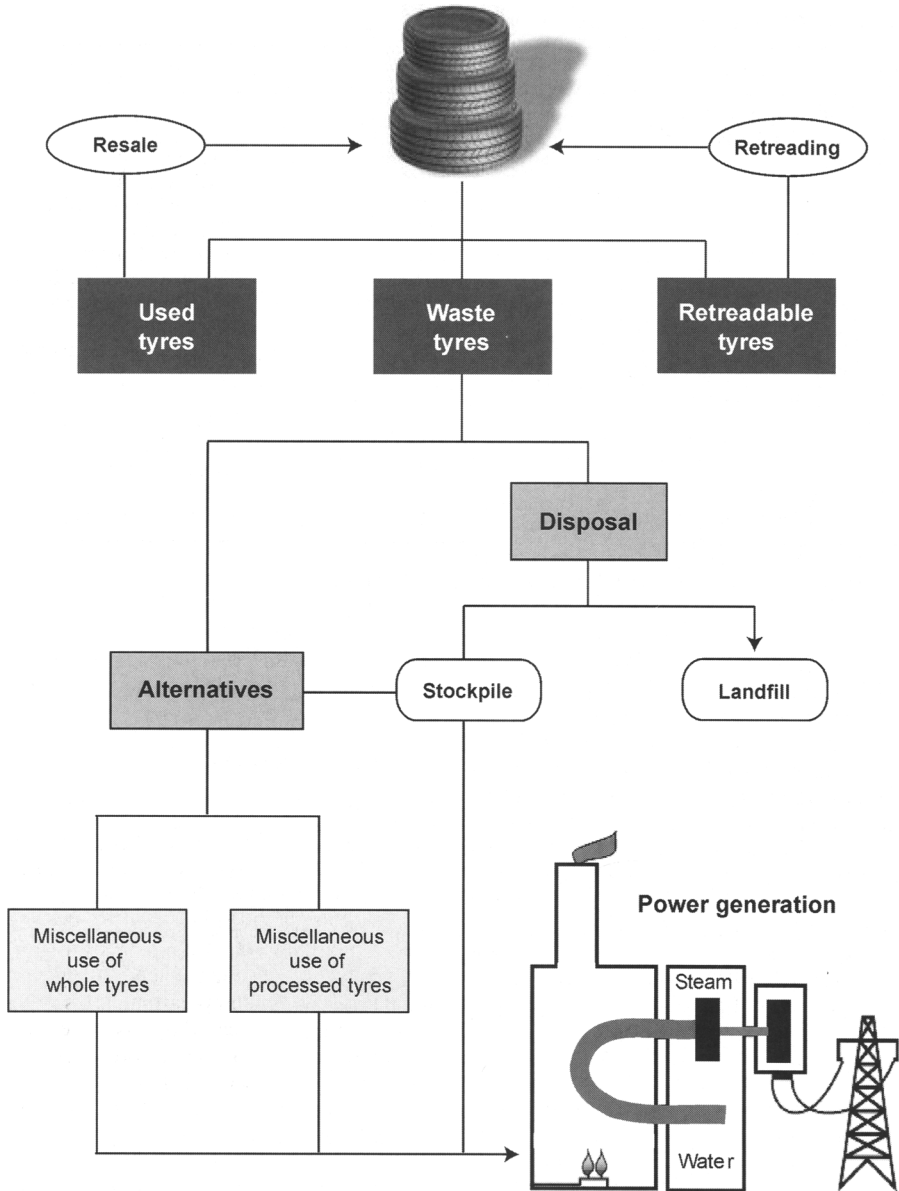


Fig. 1. Schematic diagram showing the life-cycle of tyres including current tyre waste management strategies.

literature data, we also discuss a case study of tyre co-combustion at the Purdue University power plant, where both solid waste products and atmospheric emissions have been chemically characterized.

Used and retreadable tyres

Substantial numbers of tyres are still usable on vehicles after they have been removed from

initial service. In 2001, approximately 30 million used tyres were resold either in the USA or in other countries (Table 1), whereby it is estimated that 15 million were exported, representing about 7% of the market for discarded tyres (Fig. 3b).

Another important pathway for worn tyres is recycling by retreading (Fig. 1). The estimated number of retreaded tyres in the USA ranges

Table 1. Discarded tyre disposition in 2001 for the USA*

General application	Millions of tyres	Specific application
Total worn tyres accumulated	281	
Total discarded tyres used [†]	218	
Tyre fuel	115	Total tyre fuel
	53	Cement kilns
	19	Pulp and paper mills
	18	Electric utilities
	11	Industrial boilers
	14	Tyre-to-energy facilities
Rubber products	41	Total rubber products
	33	Ground rubber
	8	Cut, punched, or stamped
Civil engineering applications	40	
Miscellaneous applications	7	
Resale	30	Total resale
	15	Export for resale
	15	Domestic resale
Retreading	16	
Disposal in landfills	28	Landfill or monofill

*Data from RMA (2002).

[†]Excludes domestic resale and retreated tyres; for details, see RMA (2002).

from 16.4 million (Table 1) to 24.2 million (TRIB 2003) for the year 2001. The consumer price for retreaded tyres may be considerably less than that of new tyres, because retreading a worn tyre requires up to 70% less oil when compared to manufacturing a new tyre (EPA 2003). Retreading relatively expensive tyres, such as truck tyres, which amount to ~15% of all discarded tyres in the USA, is an economic and widely practised option (Jang *et al.* 1998; Blumenthal 1993). Moreover, 80% of the tyres used by the commercial aviation industry are retreaded (TRIB 2003). Retreading offers the best strategy for value recovery from tyres,

requiring the least new material and energy (Amari *et al.* 1999).

Disposal of waste tyres

Landfills

In the past, landfills have been the preferred method of waste tyre disposal because burial eliminates the fire hazard associated with above-ground storage, as well as the unsightliness of tyre piles. However, problems with this method have become increasingly apparent. Because of their composition, tyres are neither decomposable, nor can they easily be compacted (Blumenthal 1993). Therefore, tyres occupy large volumes of landfills, contributing to the already prevalent problem of landfill crowding.

The large amount of empty space (75% of the volume is void; Jang *et al.* 1998), that results from the shape of tyres is often filled by air or other gases, causing the tyre to be buoyant relative to its surroundings when buried. Therefore, tyres disposed of in landfills tend to rise to the surface, disrupting the protective final cover of closed landfills (Blumenthal 1993). This disruption is potentially hazardous because precipitation can then easily enter the landfill, increasing the likelihood for contaminant leaching and subsequent transport into surrounding groundwaters or soils. Additionally, a breach in the landfill cover allows insects, birds, and rodents to interfere with landfill processes, and further

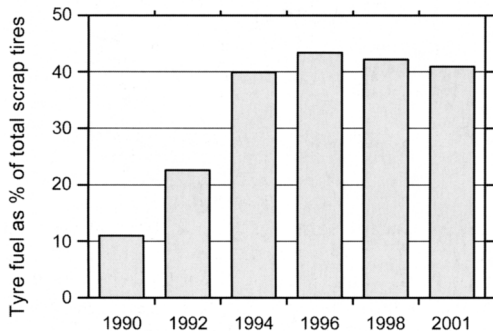


Fig. 2. Evolution of tyre fuel consumption since 1990, shown as percentage of the total amount of tyres discarded in the USA (data from RMA 2002).

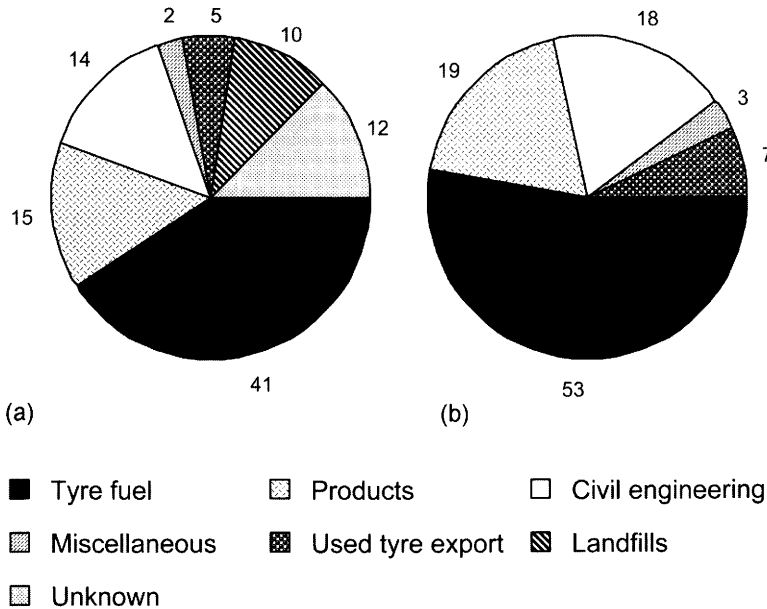


Fig. 3. Tyre data from 2001 for the USA: (a) Disposition of tyres, shown as percentage of the total number of discarded tyres (281 million tyres); (b) Markets for tyres, shown as percentage of the total number of discarded tyres used (218 million tyres) (data from Table 1).

allows for the escape of landfill gases (e.g., methane), which could otherwise be used as a source of energy. Owing to these complications, many landfill operators refuse tyre disposal or charge high tipping fees (Ohio Air Quality Development Authority 1991; EPA 1999). The regulatory trend is to ban whole tyres from landfills (Blumenthal 1993; EPA 1999), but the placement of shredded tyres in monofills has become a prominent alternative for long-term storage of scrap tyres in the recent past (RMA 2002). Today in the USA, about 10% of all worn tyres are still disposed of in landfills (Fig. 3a). Moreover, tyre shreds are now widely used in landfill engineering as well.

Stockpiles

The increasing cost or unavailability of landfill disposal as well as the existence of sham recycling operations can contribute to the growth of tyre stockpiles. The hazards associated with stockpiled tyres are equal to, if not greater than, those associated with landfill disposal. The most significant problem of tyre stockpiles is that they are prone to fires. Once tyre fires are ignited, they are extremely difficult to extinguish because of the almost continuous source of oxygen contained within the void space of the

tyres. Moreover, quenching the tyre fires with water increases the production of pyrolytic oil, which can be transported by the water into the environment, where it aggravates soil and groundwater contamination (Jang *et al.* 1998). Uncontrolled tyre burning also releases significant amounts of hazardous gases and unburned hydrocarbons, which create thick black smoke plumes (Blumenthal 1993; EPA 1997). Moreover, extinguishing fires with water creates steam clouds, which serve to entrain even more particulate matter (Lemieux, personal communication 2003).

Outdoor tyre stockpiles also represent a breeding ground for insects and rodents. Rainwater collects in the open space of the impermeable tyres, and, together with trapped windblown pollen and dust, creates an ideal environment for the development of mosquito larvae. The warm, dark recesses of the tyres also provide a supportive environment for rodent habitation. Both rodents and mosquitoes pose a potential threat to human health due to their ability to transmit various diseases (Ohio Air Quality Development Authority 1991).

As a result of stockpile abatement programmes, the number of stockpiled tyres in the USA has decreased from 700 or 800 million in 1994 to approximately 300 million at the end

of 2001 (RMA 2002). These abatement programmes take advantage of various alternative methods to tyre disposal.

Alternatives to waste tyre disposal

Several approaches have been made to reduce the number of landfilled, stockpiled, or illegally dumped tyres. Markets now exist for about 78% of all waste tyres (Fig. 3a), up from ~34.5% in 1990 (Blumenthal 1993). These markets continue to grow due to innovative approaches, which include improvements in design and material to increase the lifetime of a tyre, as well as various alternatives to tyre disposal (Table 2).

Utilization of whole waste tyres

Whole waste tyres can be utilized for many practical purposes including: crash barriers for both

automobiles and boats; breakwaters to protect shorelines and harbours; playground equipment; erosion control in mountainous areas (e.g., construction of retaining walls); and artificial reefs to encourage growth of marine populations and natural reef systems (CIWMB 1992; Jang *et al.* 1998).

Utilization of processed waste tyres

Processing of discarded tyres by shredding, cutting, stamping, or grinding is becoming an increasingly important part of the waste tyre cycle. Shredding can reduce the tyre volume by up to 75%, thus significantly reducing transportation costs (Jang *et al.* 1998). Shredded tyres are used extensively in many civil engineering applications, which in the USA consumed 40 million, or 14% of all old tyres accumulated in 2001 (Table 1, Fig. 3a). In some other countries, civil engineering applications consume most

Table 2. Alternatives to waste tyre disposal*

	Advantages	Disadvantages
Reuse, retreading	Extended tyre life Conserves resources	Limited market Eventual disposal required
Whole tyre utilization		
Bumpers, crash barriers, and erosion control	Effective Low cost	Eventual disposal required
Artificial reefs	Increase fish habitation	Expensive Potential instability
Processed tyre utilization		
Road base fill	Good filling agent	Potential leachate
Landfill operations and construction	Wide variety of applications Cost-efficient	Cutting or shredding required
Highway construction	Reduced road noise Increased durability	Current implementation limited and costly
Reclaimed rubber products	Unlimited possible products	Eventual disposal required
Pyrolysis/gasification	Energy-efficient Low air emissions Potentially saleable products	Potential for fugitive emissions Economically marginal
Tyres as energy source		
Cement kilns	Conserve natural resources Reduce NO _x emissions Complete tyre combustion Large market Ash incorporated into product	Increase in particulate emissions Increase in Zn emissions
Pulp and paper mills	Conserve natural resources	Large transportation costs for tyres Limited use Necessary to implement better emission controls Increased maintenance costs Tyres must be de-wired
Utility plants	Decrease SO ₂ and NO _x emissions	Increase in particulates released into atmosphere
Dedicated tyre-to-energy facilities	Conserve fossil fuels Use whole tyres Conserve fossil fuels	Increase Zn concentration in ash Start-up costs very high Continuous tyre supply needed

*Sources: CIWMB 1992; Amari *et al.* 1999; RMA 2002; and other references cited in text.

of the discarded tyres (e.g., Finland: 70%; Mukherjee *et al.* 2003). Shredded tyres are used for landfill construction and operation (e.g., gas venting and leachate collection systems, daily covers, cap closures); road bases; rubber-modified asphalt concrete; highway embankments; drainage systems; erosion control; and as a substitute for wood chips and gravel (Blumenthal 1993).

Another promising utilization of shredded tyres is pyrolysis, that is, the thermal degradation in the absence of oxygen. This process converts waste tyres into secondary products of significant value, for example, various chemicals, oils, and residual char, which can be used as smokeless fuel, carbon black, or activated charcoal (Williams & Taylor 1993; Mastral *et al.* 1999a). However, pyrolysis is an expensive method of tyre utilization, and the resulting oils may contain high concentrations (up to 10 wt%; Williams & Taylor 1993) of polycyclic aromatic hydrocarbons (PAH), some of which are carcinogenic and/or mutagenic (Lee *et al.* 1981). These high PAH concentrations limit the use of the tyre-derived oil as a substitute for petroleum-derived fuels, because the oil would represent a health hazard when combusted.

A large amount of scrap tyres is also processed into ground rubber, or crumb rubber, either by mechanical grinding with an abrasive at ambient temperature or by cryogenic fracturing after cooling with liquid nitrogen (EPA 1994). The most important use of ground rubber is as an asphalt modifier in highway construction. Ground rubber is mixed with asphalt to produce reinforced road paving materials. These materials, known as rubberized asphalt or asphalt rubber, are characterized by reduced road noise, shorter breaking distances, and, due to increased durability of the roadway, lower maintenance costs (CIWMB 1992; Blumenthal 1993; RMA 2002). Ground rubber can also be used to fabricate athletic field surfaces, carpet underlayment, and railroad crossing beds. Discarded tyres are further processed to reclaim rubber, yielding a raw material to make various commercial products, including new tyres. In the USA, the grinding method accounted for 15% of all discarded tyre usage in 2001, whereas products made through cutting, punching, or stamping only accounted for about 4% (Table 1).

Tyres as energy source

A promising and relatively new way to utilize both whole and processed old tyres (Fig. 1) is to recover the energy contained in this waste material. As an energy source, tyres represent an attractive alternative to other fuels due to their high heating value. On average, the

energy content of tyres ranges between 27 and 39 MJ/kg; the value varies because of the varying compositions of tyres combusted. This energy content is relatively high when compared to the average energy contained in bituminous coal, and is more than twice the heating value of municipal solid waste or chipped wood waste (Table 3). On the other hand, less than 40% of the energy contained in tyres is recoverable through combustion, as demonstrated by life-cycle energy budget calculations that consider both the energy consumed during production of the tyre and the energy recovered from their use as fuel (Amari *et al.* 1999). Another important advantage of tyre fuel is that it typically contains less than 2 wt% moisture, which is considerably less than coal and negligible compared to wood waste (Table 3). Owing to these advantages, the market for tyres as fuel has grown in the past decade (Fig. 2), and is the single most important end market for discarded tyres in the USA (Fig. 3b) and other countries, for example, Switzerland (50%; BUWAL 2001) and Japan (61% by weight; JATMA 2002). Waste tyres are a typical example of an opportunity fuel (Johnson *et al.* 1997), that is, a byproduct of other industrial processes with a significant potential in reducing the costs of generating electricity.

Although tyres can be used as an *alternate* fuel, they are most often utilized as a *supplemental* fuel. Tyre fuel exists either in shredded form (known as TDF, or tyre-derived fuel) or as whole tyres. Tyre-derived fuel consists of tyre chips, usually no larger than 5 cm on a side (Blumenthal 1993). The size reduction procedure is itself an energy-intensive process, and costs increase as the particle size decreases (Atal & Levendis 1995; Amari *et al.* 1999). The cost of cryogenic grinding of tyres can be as much as five times higher than that of pulverizing coal (Atal & Levendis 1995). Whole tyres or TDF

Table 3. Comparison of typical moisture contents and heating values of select fuels*

Fuel	Moisture (wt%)	Heating value (MJ/kg)
Waste tyres	<2	27–39
Bituminous coal	3–10	26–30
Petroleum coke	0.5–1	29–35
No. 6 fuel oil	—	42–43
Municipal solid waste	15–40	8.1–13
Chipped wood waste	10–60	9.3–13

*Adapted from CIWMB (1992), with additional data from Table 4.

are combusted together with traditional fuels, predominantly coal, in cement kilns, pulp and paper mills, and electric utility or industrial boilers (Table 1). In addition, whole tyres are used as the sole fuel source in dedicated tyre-to-energy facilities, which, however, only combusted ~12% of all tyre fuel used in the USA in 2001 (Fig. 4). Tyres may also be combusted in waste-to-energy facilities designed for the incineration of municipal solid waste (Amari *et al.* 1999).

Tyres as supplemental fuel

Cement kilns. Cement kilns use whole tyres or TDF to supplement energy generated through the combustion of coal. These facilities represent by far the most important tyre fuel market (Table 1, Fig. 4). The recovered energy is used to produce the clinker by heating a mixture of finely ground calcareous, argillaceous, and siliceous materials to 1500–1600 °C. The high flame temperature (up to 2000 °C) and long gas residence times (several seconds at >1000 °C) allow for near complete combustion of the tyres (Ohio Air Quality Development Authority 1991; Giugliano *et al.* 1999; Karell & Blumenthal 2001). Moreover, unlike in other facilities, the reinforcing steel wires do not need to be removed from the tyre prior to combustion, because they serve as a source of supplemental Fe for the cement, thus reducing raw material needs (Barlaz *et al.* 1993; Giugliano *et al.* 1999). Similarly, the ash resulting from tyre combustion is beneficial to the cement-making process, as it is incorporated into the product material rather than discarded (Blumenthal 1993; Giugliano *et al.* 1999). Despite these facts, scrap tyres cannot completely replace other fuels because of their high contents of Zn (see below), which would adversely

affect clinker quality and setting times (Karell & Blumenthal 2001).

Pulp and paper mills. The second largest consumers of tyre fuel are pulp and paper mills (Table 1, Fig. 4). Because the production of pulp and paper is an energy-intensive process, these facilities typically have their own boilers and turbines to meet electrical needs. Pulp and paper mills use TDF instead of whole tyres to supplement wood waste as fuel. The wood waste, also referred to as hog fuel, consists of chipped bark and other unusable tree parts (Barlaz *et al.* 1993). The use of TDF helps maintain constant combustion conditions in the stoker grate boiler system, which are not easily achieved using wood waste alone (Ohio Air Quality Development Authority 1991). The fuel-feeding process of these types of boilers, however, requires that the TDF be almost entirely free of wires (Jones *et al.* 1990). This requirement increases the costs of fuel significantly.

Electric utility and industrial boilers. The use of tyres in electric utility and industrial boilers comprises ~26% of the market for tyre fuel (Fig. 4). Although whole tyres can be used in some industrial boilers, most facilities combust TDF mixed with coal in a variety of concentrations ranging from as little as 2 to as much as 40% TDF by weight (e.g., Tesla 1994; EPA 1997; Ohio Air Quality Development Authority 1991). There are many different types of boilers that can utilize TDF combustion to augment energy generation (see Amari *et al.* 1999), the most popular of which is the cyclone-fired boiler. This type of boiler typically requires TDF pieces to be smaller than 2.5 cm × 2.5 cm, and the tyre chips must be de-wired prior to combustion (Ohio Air Quality Development Authority

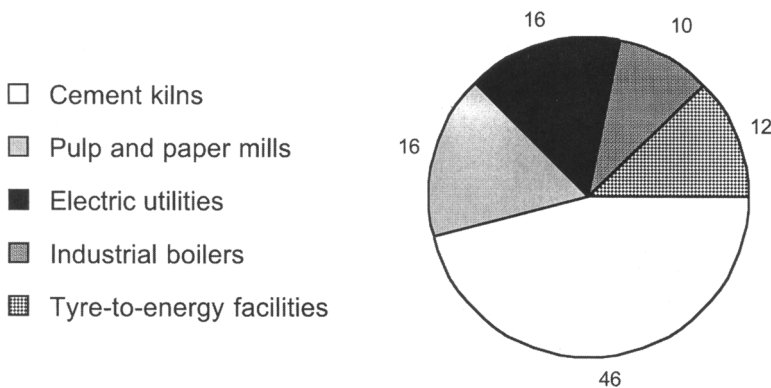


Fig. 4. Markets for tyre fuel consumed in the USA during 2001 (data from Table 1).

1991). De-wiring increases the costs of energy recovery by 25–50% (Amari *et al.* 1999). Some power companies have also conducted successful test burns with crumb-rubber TDF in their pulverized-coal boilers (Jones 1998).

Tyres as alternate fuel

The most direct process to use tyres for energy recovery is to combust whole tyres. Facilities that combust discarded tyres as the sole energy source are termed dedicated tyre-to-energy facilities. This type of facility is relatively new and appears to offer a promising and environmentally more suitable method for recovering energy from waste tyres. Only two such facilities are currently in operation in the USA, due to the complexity and cost associated with the construction and operation of these plants (RMA 2002). Nevertheless, combustion in tyre-to-energy facilities accounts for 12% of the end market of discarded tyres. At full capacity, one of the US facilities (in Stirling, Connecticut) combusts about 30 000 tyres daily, which amounts to more than 10 million tyres per year (Makansi 1992).

Composition of tyres

The production of modern car tyres uses more than 100 raw materials, most of which are based on petroleum products. Tyres consist of natural and synthetic rubber, typically styrene-butadiene (SBR); reinforcing fillers (e.g., carbon black, silica, clay, calcium carbonate); reinforcing fibres

(e.g., nylon, polyester, rayon, steel wires); extenders or softeners (e.g., petroleum process oils); vulcanizing agents and accelerators (e.g., zinc oxide, stearic acid, organic sulphur compounds); and various antioxidants (JATMA 2002). The amounts of these materials vary widely among manufacturers and types of tyres, but a typical passenger car tyre contains, in addition to natural and synthetic rubber, ~3–5 wt% textile fibres, 10–17 wt% steel wire, 1 wt% ZnO, and 21–31 wt% carbon black (Amari *et al.* 1999; BUWAL 2001; JATMA 2002). As a result of these rather large ranges, the chemical composition of tyres is highly variable, thus complicating the quantitative assessment of the environmental impact of tyre combustion.

A typical automobile tyre consists of several parts (Fig. 5). The tread and sidewall make up the outside of the tyre. The tread is the portion of the tyre that comes into contact with the road, whereas the sidewall, or wall, is the side of the tyre that connects the tread and the bead. The tread is reinforced from underneath by the belt, which comprises a set of fabrics and/or wires. The plies, found in association with the belt, are layers of rubber-coated cords, which are typically made of fabric, polymer, fibreglass, or steel. The set of belts and plies is often collectively referred to as the cord, or cord body, of the tyre. The cords of the tyre wrap around the bead, the part of the tyre that rests on the tyre rim. The bead consists of both rubber bead filler and a series of steel bead wires. The last major part of the tyre, the innerliner, is a low-permeability

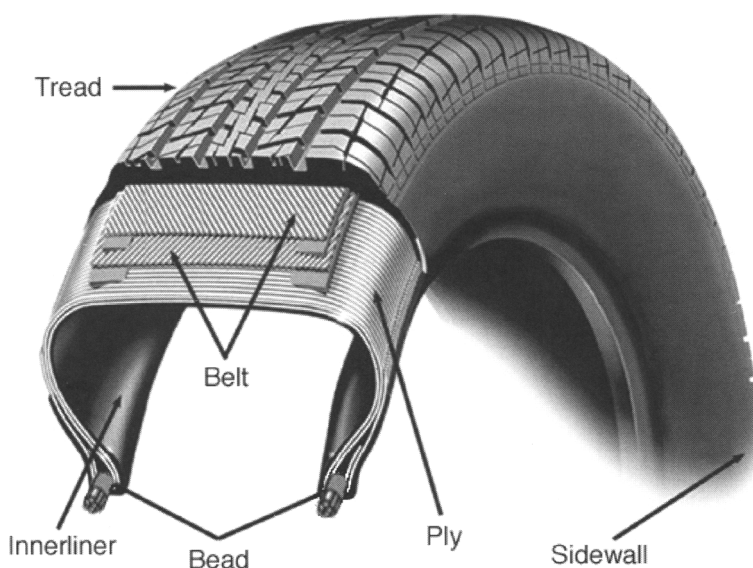


Fig. 5. Schematic diagram showing the most important parts of a car tyre. Modified from original diagram posted on the World Wide Web (<http://tiresafety.com>).

rubber. This rubber is laminated to the inside of tubeless tyres in order to improve the tyres' air retention capabilities.

The ash content of tyres and TDF varies within a large range, depending on composition. Steel-belted tyres exhibit considerably higher ash contents than textile-based tyres (Table 4). The available data show that, compared to typical coal, tyres and TDF contain significantly less N and, in general, less moisture (see also Table 3). On the other hand, tyres are distinctly richer in volatiles than coal, and their S content is similar to that of medium-S coal. Only a few studies report Cl concentrations, but these indicate that waste tyres are generally richer in Cl than typical coal (Tables 4, 5). Most investigations on waste tyres, unfortunately, do not provide analytical data for elements other than those shown in Table 4. Table 5 lists available chemical compositions of scrap tyres. The data reveal that tyres may contain considerable quantities of trace elements, in addition to the high amounts of Zn and, for steel-belted tyres, Fe (14–16 wt%; Makansi 1992; BUWAL 2001). Of particular interest is the relatively volatile element Zn, whose concentration ranges mostly between 1 and 2 wt% (Polasek & Jervis 1994), but may be as high as 4.3 wt% in certain truck tyres (Smolders & Degryse 2003). Because of the high Zn content of tyres, there is concern about diffuse Zn contamination of the environment from tyre wear. Several studies, for example, have documented steep Zn concentration gradients in roadside soils (see discussion in Smolders & Degryse 2003).

Table 5 further documents that major differences in elemental concentrations do not only exist between different types of tyres, but also between the tread and the wall of a tyre. Most of the elements are preferentially contained in the tyre walls relative to the tread. The wall is particularly enriched in Al and Ti, as well as in the volatile elements Cl, Br (in truck tyres), and Sb (Table 5). Tyre fuel consists most likely predominantly of wall material, and therefore, it is expected that tyre fuel will exhibit concentrations at the upper end of the range listed in Table 5. Of further note are the environmentally critical and volatile elements Hg and Pb, which range between 8 ppb and 0.4 ppm and 5 ppb and 65 ppm, respectively.

Solid waste products from tyre combustion

A large portion of combustion products from industrial and utility operations is regarded as waste, which is directly disposed of in surface impoundments and landfills. The interest in the chemical composition of combustion products

is largely due to concerns about the potential release of toxic elements through leaching processes in ash disposal sites (see chapters by Tishmack & Burns, Spears & Lee, and Donahoe in this volume). It is therefore necessary to know the characteristics of the discarded combustion products in order to assess possible environmental impacts. The following section concentrates on the bulk chemical composition of the solid combustion products generated by burning of pure tyres or blends of tyres and coal.

Solid waste from pure tyre combustion

The ash generated through combustion of pure tyres, that is, without an additional fuel, varies in composition depending on the kind and manufacturer of the tyre, and on the combustion conditions. Unfortunately, there are only very few chemical analyses available for pure tyre ash. To our knowledge, the only investigations reporting TDF or tyre ash compositions are those of Granger & Clark (1991) and Polasek & Jervis (1994). These authors have shown that one of the principal TDF or tyre ash components is ZnO (Table 6), which is added during the vulcanization process. Polasek and Jervis (1994) stated that ZnO may account for as much as 75 wt% of the ash, but did not publish this specific analysis.

Other major constituents of TDF and tyre ash include Al_2O_3 , SiO_2 , CaO , TiO_2 , and Fe_2O_3 (Table 6). Polasek & Jervis (1994) observed that the concentrations of Al_2O_3 and TiO_2 may be as high as 23 wt% and 33 wt%, respectively. Granger & Clark (1991) reported SO_3 contents of 15 wt% in ash produced through combustion of de-wired TDF (Table 6).

Solid waste from tyre co-combustion

As more industrial facilities are beginning to use tyres or TDF as a supplemental fuel, studies of the chemical composition of the solid waste resulting from co-combustion of tyres or TDF with other fuels (e.g., coal, natural gas, wood waste, or municipal solid waste) are becoming increasingly important. Owing to the large compositional variability observed for both tyres (Tables 4 and 5) and other fuels, it is difficult to compare data sets collected at different combustors. Direct comparison of various data sets is further complicated by differences in combustion conditions and ash collection procedures at different industrial and utility facilities. These difficulties are already a problem when comparing the chemical composition of pure coal combustion products generated at different facilities.

Table 4. Composition of various tyres, TDF, and coal

Fuel	Specifics	Source	Energy content (MJ/kg)	Moisture (wt%)	Ash (wt%)	Volatiles (wt%)	H (wt%)	C (wt%)	Fixed C (wt%)	N (wt%)	O (wt%)	S (wt%)	Cl (ppm)
Tyre	Fibreglass	Pope 1991	32.5	—	11.7	—	6.6	75.8	—	0.2	4.4	1.29	—
Tyre	Steel-belted	Pope 1991	26.7	—	25.2	—	5.0	64.2	—	0.1	4.4	0.91	—
Tyre	Nylon	Pope 1991	34.6	—	7.2	—	7.0	78.9	—	<0.1	5.4	1.51	—
Tyre	Polyester	Pope 1991	34.3	—	6.5	—	7.1	83.5	—	<0.1	1.7	1.20	—
Tyre	Kevlar-belted	Pope 1991	39.2	—	2.5	—	7.4	86.5	—	<0.1	2.1	1.49	—
Tyre	De-wired	Mastral <i>et al.</i> 2000	38.6	0.9	3.8	67.3	8.3	88.6	31.1	0.40	—	1.40	—
Tyre	Car	Karell & Blumenthal 2001	36.9	—	3.9	—	7.6	89.5	—	0.27	<0.01	1.88	700
Tyre	Truck	Karell & Blumenthal 2001	34.8	—	5.5	—	7.5	89.6	—	0.25	<0.01	2.09	600
Tyre	—	Burger 1991	33.7	0.8	10	65	—	78	—	0.2	—	1.4	1500
TDF	—	Karell & Blumenthal 2001	36.5	—	4.2	—	7.6	89.5	—	0.27	<0.01	1.92	700
TDF	De-wired	Karell & Blumenthal 2001	36.0	0.6	4.8	66.6	7.1	83.9	28.0	0.24	2.2	1.23	1490
TDF	Pulverized	Biagini <i>et al.</i> 2002	36.9	0.9	5.1	63.7	7.0	78.9	30.2	0.6	5.6	1.85	—
TDF	1.25 cm, rubber 'fuzz'	Granger & Clark 1991	32.1	2.3	16.5	—	6.3	69.7	—	0.45	3.4	1.30	700
TDF	5 cm, de-wired	Granger & Clark 1991	32.6	1.0	8.7	—	6.7	72.2	—	0.36	9.7	1.23	900
TDF	5 cm, with metal	Granger & Clark 1991	31.1	0.8	23.2	—	5.8	67.0	—	0.25	1.6	1.33	300
TDF	With metal	Gaglia <i>et al.</i> 1991	37.8	0.6	4.8	—	7.1	83.9	—	0.24	2.2	1.23	—
TDF	Crumb rubber	Hutchinson <i>et al.</i> 1991	37.9	0.2	4.2	64.8	—	—	28.9	—	—	1.90	—
TDF	2.5 cm, with metal	Hutchinson <i>et al.</i> 1991	34.6	2.4	13.2	60.7	—	—	22.4	—	—	1.30	—
TDF	2.5 cm, de-wired	Hutchinson <i>et al.</i> 1991	36.5	1.3	6.8	—	—	—	—	—	—	1.40	—
TDF	1.25 cm, de-wired	Hutchinson <i>et al.</i> 1991	36.2	0.8	9.5	60.9	—	—	27.3	—	—	1.50	—
TDF	3.75 cm, with metal	Howe 1991	29.2	8.6	14.8	54.3	6.0	62.0	22.4	0.25	7.3	1.19	—
TDF	1.88 cm, with metal	Phalen <i>et al.</i> 1991	36.7	0.4	8.3	62.7	6.8	77.4	28.6	0.37	5.1	1.62	—
TDF	—	Jones <i>et al.</i> 1990	36.0	0.6	4.8	—	7.1	83.9	—	0.24	2.2	1.23	—

TDF	Crumbs, <0.64 cm, de-wired	Lemieux 1994; Miller <i>et al.</i> 1998	37.3	0.8	7.2	65.5	7.2	76.0	26.4	0.34	7.1	1.75	3100*
TDF	Ground	Levendis <i>et al.</i> 1998a	29.0	—	26.0	52.3	5.3	60.9	21.7	0.28	7.1	2.46	—
TDF	2.5 cm, not de-wired	Unpublished data [†]	—	—	—	—	—	83.3	—	—	—	1.69	2938
TDF	—	University of Iowa [‡]	33.6	6.6	4.7	60.4	6.6	77.1	28.4	0.50	3.2	1.39	—
Coal	Sub-bituminous	Unpublished data [§]	25.9	14.6	7.2	33.6	—	75.4	43.9	—	—	1.47	215
Coal	Bituminous	Levendis <i>et al.</i> 1998a	29.2	—	13.7	34.4	4.7	71.9	51.9	1.36	7.0	1.36	—
Coal	Bituminous	Karell & Blumenthal 2001	31.5	—	7.8	—	5.1	75.8	—	1.50	8.2	1.60	—
Coal	Bituminous	Hower <i>et al.</i> 2001	28.4	2.7	14.0	38.4	4.8	64.3	—	1.43	—	4.37	—
Coal	—	University of Iowa [‡]	28.1	8.5	9.4	32.4	4.6	67.7	49.7	1.37	7.3	1.14	—

*Represents total halogens, calculated as Cl.

[†]TDF used in test burn conducted by the Purdue University power plant (this study).[‡]Data compiled from original University of Iowa data set provided by Schwarzhoff & Milster (personal communication 2003).[§]Illinois Basin coal from Southern Indiana, used in Purdue University power plant test burn (this study).

However, test burns that were conducted at a single facility to study the effects of adding TDF to a base fuel reveal chemical trends that may be similar to those observed at other facilities, and thus could be useful for assessing ash disposal and groundwater protection strategies.

In experiments to study the potential for emissions of hazardous air pollutants from combustion of TDF, Lemieux (1994) co-fired natural gas and wire-free crumb rubber at ~900–1000 °C in a rotary kiln incinerator simulator (composition of crumb rubber given in Tables 4 and 5). The resulting fly ash is characterized by high contents of Al₂O₃, SiO₂, CaO, and ZnO (Table 6). The solid combustion product is thus similar to the ash produced through combustion of pure tyres or TDF (cf. data of Granger & Clark 1991 and Polasek & Jervis 1994 in Table 6), documenting that co-firing TDF with natural gas does not affect the ash composition to any significant extent. Using X-ray diffractometry, Lemieux (1994) identified cristobalite/quartz (both SiO₂), mullite (Al₆Si₂O₁₃), and willemite (ZnSiO₄) as major phases in this fly ash.

Table 7 lists the results of test burns at two coal-combusting power plants: a western Kentucky power plant, which burned pure Illinois Basin coal (Table 4) and two blends of this coal + TDF (99 wt% coal + 1 wt% TDF; 97 wt% coal + 3 wt% TDF) in a cyclone boiler (Hower *et al.* 2001); and the Purdue University power plant, which used a stoker boiler to combust a comparable Illinois Basin coal as well as a blend containing 95 wt% of this coal and 5 wt% TDF (Table 4; this study).

The Kentucky data (Hower *et al.* 2001) reveal considerable changes in the concentration of several components in the bulk fly ash with the addition of TDF. The most drastic changes are observed for Zn and Pb in the fly ash resulting from combustion of the 3 wt% TDF blend: relative to pure coal, the Zn content of the fly ash more than doubled, whereas that of Pb increased from undetectable levels to more than 300 ppm (Table 7). Notable increases in concentration are also observed for Na₂O, Al₂O₃, SiO₂, and TiO₂, whereas most other components of the bulk fly ash, including Cr, Ni, and Cu, show reduced concentrations (Fig. 6). For the bulk bottom ash samples, pronounced increases in concentration were only observed for CaO and Fe₂O₃. When compared to the pure coal and the blend with 3 wt% TDF, there are some inconsistencies in the chemical trends observed for the blend containing 1 wt% TDF (Table 7); these may result from heterogeneities in the fuel.

The Purdue University data (this study) revealed that the bulk fly ash from the coal +

Table 5. Concentrations of various elements (in ppm) in select tyres or parts thereof, and in TDF

	Typical car tyre tread		Typical car tyre wall		Typical truck tyre tread		Typical truck tyre wall		Bulk car tyre	TDF	TDF (de-wired)	TDF (de-wired)	Car tyre	TDF (not de-wired)	
	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.	Mean	Std. dev.						Mean	Std. dev.
Be	—	—	—	—	—	—	—	—	—	0.4	—	—	—	—	<0.5
Na	610	160	330	85	510	140	530	130	—	—	—	—	—	—	<70
Mg	32	—	230	160	44	43	688	670	—	—	—	—	—	—	500
Al	420	120	5800	1500	340	90	33000	9000	—	—	—	—	—	—	2520
Cl	690	220	1300	420	410	140	1900	690	—	318	—	—	—	—	2938
K	180	70	520	130	1700	360	—	—	—	—	3100	1490	—	—	332
Ca	330	240	650	440	160	100	2000	1000	—	—	—	3780	—	—	1550
Sc	0.011	0.008	0.6	0.1	—	—	2.2	0.4	—	—	—	—	—	—	<10
Ti	195	84	3500	1200	42	23	64000	18000	—	—	—	—	—	—	1480
V	0.8	0.4	4.8	1.6	0.9	0.3	12	6	—	0.4	—	—	—	—	<5
Cr	4	2	8	4	3.8	1.7	22	13	90	1	<5	97	0.025	—	<68
Mn	1.9	—	7.4	2.3	2.9	1.1	5.5	4.2	—	11	—	—	—	—	<80
Fe	210	70	790	210	—	—	1300	300	160000	1605	295	3210	—	—	4400
Co	2.9	0.8	2.4	0.7	2.6	0.4	1.9	0.6	—	182	—	—	—	—	63
Ni	—	—	—	—	—	—	—	—	80	2	—	—	0.013	—	22
Cu	—	—	—	—	—	—	—	—	200	70	—	—	—	—	38
Zn	17000	3000	20000	4000	12000	1400	19000	4000	8030	12137	21400	15200	15300	—	10900
As	—	—	0.6	0.3	—	—	—	—	—	0.8	—	—	0.65	—	0.9
Se	—	—	—	—	—	—	—	—	—	3.6	—	—	—	—	<0.1
Br	1.8	1	3	1.6	2	1.5	1400	560	—	—	—	—	—	—	—
Mo	—	—	—	—	—	—	—	—	—	1.5	—	—	—	—	<2
Cd	—	—	—	—	—	—	—	—	10	0.4	<5	6	<2	—	0.1
In	—	—	—	—	—	—	—	—	—	10.8	—	—	—	—	—
Sb	0.06	0.06	46	15	0.18	0.9	11	3.3	—	3.4	—	—	—	—	3.6
Ba	17.2	—	17.3	—	—	—	—	—	—	0.4	—	—	—	—	11.6
Sm	0.1	0.03	0.4	0.1	0.07	0.03	1.2	0.3	—	—	—	—	—	—	0.2
Hg	—	—	—	—	—	—	—	—	—	0.4	—	—	<0.1	—	0.008
Pb	—	—	—	—	—	—	—	—	50	38	51	65	0.005	—	6
Source	Polasek & Jervis 1994		Polasek & Jervis 1994		Polasek & Jervis 1994		Polasek & Jervis 1994		BUWAL 2001	Hutchinson <i>et al.</i> 1991	Lemieux 1994; Miller <i>et al.</i> 1998	Karell & Blumenthal 2001	Mukherjee <i>et al.</i> 2003	Unpublished data*	

* Average of three analyses of TDF, used in test burn conducted by the Purdue University power plant (this study).

Table 6. Composition of TDF and tyre combustion ash

	Ash from TDF rubber 'fuzz' (1.25 cm)*	Ash from TDF (de-wired, 5 cm)*	Ash from TDF (with metal, 5 cm)*	Ash from typical car tyre [†]		Ash from wire-free crumb rubber (<0.64 cm) [‡]	
				Mean	Std. dev.	Sample TB3	Sample TB6
<i>Major element oxides (wt%)</i>							
Na ₂ O	1.07	1.10	0.13	—	—	0.92	1.6
MgO	0.73	1.35	0.10	0.5	0.2	1.4	1.6
Al ₂ O ₃	6.99	9.09	1.93	18	3	5.3	3.6
SiO ₂	18.21	22.00	5.16	—	—	68	68
P ₂ O ₅	0.56	1.03	0.10	—	—	0.002	0.0002
SO ₃	8.35	15.38	0.99	—	—	0.001	0.001
K ₂ O	0.55	0.92	0.14	0.5	0.2	0.70	0.70
CaO	5.99	10.64	0.56	0.4	0.2	4.3	4.0
TiO ₂	6.01	2.57	0.14	28	5	0.25	0.14
Fe ₂ O ₃	30.93	1.45	0.35	1.0	0.1	1.2	1.2
ZnO	20.60	34.50	5.14	22	2	3.0	6.5
Metal [§]	—	—	85.28	—	—	—	—
<i>Trace elements (ppm)</i>							
Cl	—	—	—	86	43	—	—
Sc	—	—	—	7.2	0.9	—	—
V	—	—	—	46	9.9	—	—
Cr	—	—	—	107	26	20	100
Mn	—	—	—	160	18	—	—
Co	—	—	—	19	3	50	100
Ni	—	—	—	—	—	70	30
Cu	—	—	—	—	—	20	1
Br	—	—	—	280	77	—	—
Sr	—	—	—	—	—	20	100
Zr	—	—	—	—	—	100	100
Sb	—	—	—	7	1	—	—
La	—	—	—	33	5	—	—
Sm	—	—	—	3.8	0.6	—	—
Pb	—	—	—	—	—	10	9

*Data from Granger & Clark (1991). Major element oxides normalized to 100 wt%. Ash represents residue from combustion of pure TDF, whose fuel characteristics are listed in Table 4.

[†]Data from Polasek & Jervis (1994), for pure tyre ash. Major element oxides add up to only 71 wt%; no further details available.

[‡]Data from Lemieux (1994), collected after co-firing natural gas and TDF crumbs. Major element oxides add up to 85 and 87 wt%, respectively. Ash represents fly ash from combustion of TDF crumbs, whose fuel characteristics are listed in Tables 4 and 5. Lemieux (1994) suspected that some elements (Al, Si, Zr) may have originated from the rotary kiln insulation.

[§]Metal (reported as Fe₂O₃) was removed from sample before the analysis was performed.

TDF mixture was distinctly richer in SO₃, Zn, Ge, and As than the fly ash from pure coal. Most pronounced are the changes in the contents of SO₃ and Zn, which increased by factors of more than 2 and 16, respectively (Fig. 6). This result can be explained by the average concentrations of these volatile components in the two fuels used in the test burn: the coal + TDF blend contains 2.1 wt% S and 183 ppm Zn, whereas the pure coal only contains 1.5 wt% S (Table 4) and 36 ppm Zn (Gieré, unpublished data). The levels of most other analysed trace metals in the bulk fly ash decreased with the addition of TDF. For the bottom ash, many of

the patterns observed for the fly ash are reversed (Table 7).

Overall, the Kentucky and Purdue studies show similar trends. The most important change resulting from the addition of TDF to coal is the increase in the Zn content of the fly ash. In contrast, most other trace elements have lower concentrations in the fly ash resulting from the combustion of the blend. Despite many similarities, two notable differences between the two test burns are observed. SO₃ and Pb showed opposite trends in the Purdue fly ash compared to the Kentucky fly ash. In both studies, enrichments of many trace elements

Table 7. Comparison of bulk solid waste compositions from two coal-combusting facilities

Fuel type (wt%) Boiler type Combustion temperature Ash type*	Hower <i>et al.</i> 2001		Hower <i>et al.</i> 2001		Hower <i>et al.</i> 2001		Purdue University		Purdue University					
	100% Coal Cyclone 1649 °C	FA n = 1	99% Coal + 1% TDF Cyclone 1649 °C	BA n = 1	FA n = 1	97% Coal + 3% TDF Cyclone 1649 °C	BA n = 1	FA n = 1	100% Coal Stoker 1500 °C	BA n = 1	FA† n = 2	95% Coal + 5% TDF Stoker 1500 °C	BA n = 3	FA† n = 2
C (wt%)	0.63	1.40	1.79	21.36	3.17	0.81	2.0	15.2	2.1	0.6	16.2	0.6	16.2	
Major element oxides (wt%), normalized to 100 wt% on a C-free basis														
Na ₂ O	0.12	1.04	0.38	6.94	0.42	2.10	0.29	0.86	0.26	0.02	0.79	0.02	0.79	
MgO	0.67	0.46	0.64	0.32	0.60	0.19	0.56	0.64	0.51	0.02	0.69	0.02	0.69	
Al ₂ O ₃	20.90	17.16	20.02	21.72	18.95	20.61	29.3	23.2	27.3	0.09	22.4	0.09	22.4	
SiO ₂	48.07	29.26	44.38	30.26	44.36	35.14	47.9	42.8	46.2	1.2	41.3	1.2	41.3	
P ₂ O ₅	0.27	0.17	0.24	0.02	0.22	0.05	0.13	1.48	0.16	0.03	1.47	0.03	1.47	
SO ₃	0.41	1.04	0.57	0.37	0.79	0.79	0.32	7.65	0.8	0.1	16.4	0.1	16.4	
K ₂ O	2.71	3.11	2.75	2.82	2.80	3.18	1.74	2.67	1.52	0.06	2.58	0.06	2.58	
CaO	4.85	8.87	5.25	2.19	5.93	3.48	0.54	1.03	0.55	0.03	1.30	0.03	1.30	
TiO ₂	1.08	1.12	1.13	1.40	1.10	1.45	1.22	1.68	1.25	0.02	1.72	0.02	1.72	
Fe ₂ O ₃	20.92	37.76	24.63	33.96	24.84	33.00	18.0	18.0	21.4	1.6	11.2	1.6	11.2	
Trace elements (ppm) [‡]														
V	702	174	233	410	252	410	281	614	307	6	548	6	548	
Cr	283	125	156	212	80	212	220	487	262	70	382	70	382	
Mn	179	199	99	135	262	135	100	200	152	50	150	50	150	
Ni	397	67	244	233	62	233	358	1765	291	22	1185	22	1185	
Cu	298	39	166	230	52	230	118	945	118	9	680	9	680	
Zn	2420	90	1840	5750	275	5750	68	3475	270	28	56400	28	56400	
Ga	107	<0.6	46	98	<0.6	98	20	587	19	1	411	1	411	
Ge	278	13	149	265	<0.9	265	19	1220	23.5	0.3	1925	0.3	1925	
As	468	<1	627	302	9	302	2	650	4	5	700	5	700	
Rb	<0.7	70	<0.7	<0.7	160	<0.7	91	124	83	7	107	7	107	
Sr	275	259	183	238	146	238	600	574	671	89	476	89	476	
Y	<1.2	<1.2	<1.2	<1.2	<1.2	<1.2	92	111	80	2	85	2	85	
Zr	340	192	126	230	211	230	242	222	234	8	195	8	195	
Mo	<1.3	<1.3	<1.3	<1.3	<1.3	<1.3	8	112	8	1	81	1	81	
Pb	<1.8	20	<1.8	327	<1.8	327	22	2045	31	10	1605	10	1605	

*BA and FA represent bottom ash and fly ash, respectively.

†Purdue University FA collected from an electrostatic precipitator hopper.

‡Detection limits for the Hower *et al.* (2001) data set extracted from Wong & Robertson (1993).

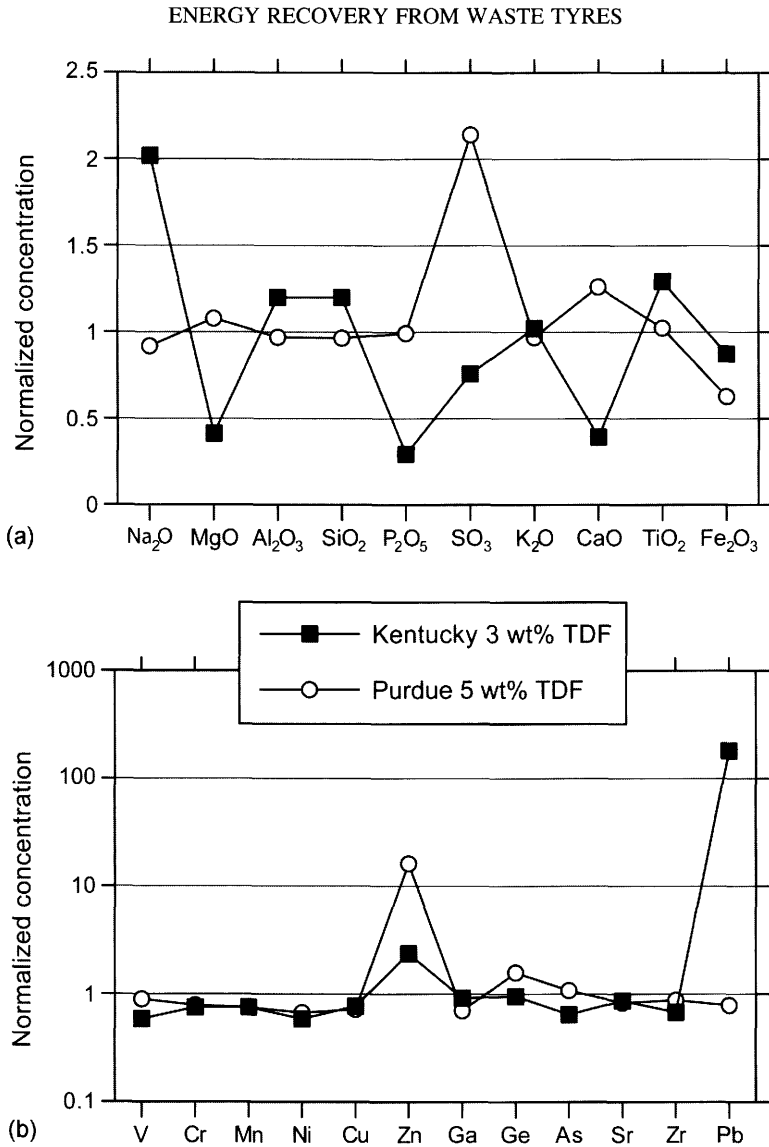


Fig. 6. Comparison of normalized concentrations of chemical components of fly ash from two boilers. Values represent concentrations in fly ash from combustion of coal/tyre blends normalized to the respective concentrations in fly ash from combustion of pure coal (data from Table 7).

are observed in the fly ash compared to the bottom ash (Table 7). This result is not surprising, however, because the bottom ash accumulates directly from the boiler, which is much hotter than the collection points for the fly ash, and therefore, the relatively volatile elements (e.g., S, Zn, As, and Pb) are partitioned into the fly ash due to temperature. Moreover, the smaller particles of the fly ash are more effective at capturing volatile elements than the larger ones

in the bottom ash, because they exhibit relatively large surface areas (e.g., Hower *et al.* 2001).

Atmospheric emissions from tyre combustion

Emissions from pure tyre combustion

As discussed above, the chemical composition of tyres differs greatly among various

manufacturers, tyre types, and tyre parts. Therefore, results of emission tests from pure tyre combustion may vary between different studies. However, several general trends in emissions have been reported from laboratory-scale studies at carefully controlled conditions: compared to coal combustion, burning of pure tyres under the same conditions typically produces much smaller amounts of NO_x ($\text{NO} + \text{NO}_2$) emitted into the atmosphere, but higher emissions of CO, PAHs, and submicrometre particulate matter (Levendis *et al.* 1996, 1998a, b; Courtemanche & Levendis 1998; Mastral *et al.* 1999b, 2000). The quantitative effects, however, vary greatly and depend on various factors, including fuel composition and particle size, boiler type, and conditions of combustion.

The influence of fuel composition on atmospheric emissions is best documented for CO_2 , SO_2 , and NO_x . In experiments using an electrically heated drop-tube furnace, Levendis *et al.* (1998a, b) observed that the CO_2 yields from tyre combustion were considerably lower than those from coal combustion, reflecting the difference in C contents of the specific fuels used (60.9 wt% vs. 71.9 wt% C, respectively; see Table 4). Moreover, the particular batch of tyres used by these authors had an S content of 2.46 wt%, which is unusually high for tyres (see Table 4), and generated higher SO_2 emissions than the bituminous coal reference material (1.36 wt% S). Finally, the NO_x emissions generated during combustion of tyres are much lower than those resulting from coal combustion because tyres typically contain considerably less N than coal (Table 4). Despite this fact, a larger fraction of the fuel-bound N is converted to NO_x when tyres are combusted rather than coal (Levendis *et al.* 1996). This conversion, however, does not compensate for the lower N content of tyres and thus, the NO_x emissions from tyre combustion remain low relative to those of coal. The influence of fuel particle size has been studied by Levendis *et al.* (1996), who showed that tyre particles combust considerably faster than coal particles of similar size.

Another key factor determining the emission levels is the bulk equivalence ratio (Φ) in the furnace, a parameter that describes the relative amounts of fuel and air. Levendis and co-workers (1996, 1998b) have shown that for both coal and tyre (and for various blends thereof) the specific emissions, that is, the mass of a certain pollutant relative to the mass of fuel burned, vary systematically when the combustion environment changes from fuel-lean (low Φ) to fuel-rich (high Φ) conditions. Their results document that, at a given temperature,

the specific emissions of NO_x , SO_2 , and CO_2 decrease moderately to strongly with increasing Φ , whereas those of CO and PAH increase. The observed changes are drastic for CO and PAH, whose specific emissions were insignificant at fuel-lean conditions, but increased exponentially with increasing Φ in fuel-rich conditions.

Emission levels are also strongly influenced by the gas temperature in the furnace. Between 1000 and 1300 °C, for example, an increase in temperature leads to a decrease in emissions of PAH, but to an increase in CO and NO_x emissions (Levendis *et al.* 1998a; Courtemanche & Levendis 1998). The observed increase in CO has been attributed to partial oxidation of hydrocarbons. Temperature-dependence of PAH emissions has also been reported below 1000 °C: in laboratory experiments using an atmospheric fluidized-bed reactor, Mastral *et al.* (1999b, 2000) have shown that within the studied range (650–950 °C), total PAH emissions were highest at 750 °C and 950 °C for coal and tyres, respectively, whereby the temperature effect was much more pronounced for the tyre fuel. In both cases, the main contribution to the total PAHs was from the most volatile compounds, that is, those consisting of three (coal) or three and four (tyre) rings. During tyre combustion, most of the emitted PAHs are associated with particulate matter, especially at higher temperatures, whereas coal combustion more frequently produces PAH emissions in the gas phase (Mastral *et al.* 1999b).

The PAH emissions from combustion processes are a particular concern because they are organic compounds, which can be more dangerous than inorganic compounds, even though they are emitted in smaller quantities. During the combustion process, PAHs are formed by two mechanisms. Pyrolysis first leads to partial fragmentation of organic compounds, producing small compounds, mainly free radicals, which are highly reactive and thus only exist for a short period of time. Subsequent pyrosynthesis recombinates these intermediary compounds into the relatively stable PAHs, which consist of two or more aromatic rings (Mastral *et al.* 1999b). One cause for the high PAH emissions from tyre combustion, relative to coal combustion, may be the large amount of carbon black in tyres, which, when only partially burned, appears to provide particle surfaces favourable for PAH deposition (Mastral *et al.* 1999b).

A larger-scale tyre combustion experiment was performed by Lemieux & Ryan (1993) in order to collect emission data from a simulated open waste tyre fire. In addition to identifying a large number of organic compounds, including

PAHs, in the emitted gases, these authors also studied particulate matter, in which they observed elevated levels of Zn and Pb.

Uncontrolled open fires in tyre dumps may burn for several months (e.g., Rhinehard tyre fire in Winchester, Virginia), generating many hazardous products of incomplete combustion, which are released directly into the atmosphere (EPA 1997). Owing to safety concerns and other factors (e.g., meteorological conditions, fire-fighting activities), it is difficult to collect air samples from the smoke plumes of these fires. The available field data revealed potentially hazardous levels of several PAHs and CO, as well as relatively high concentrations of Pb and Zn in such plumes (Ohio Air Quality Development Authority 1991; EPA 1997).

Emissions from tyre co-combustion

Besides examining the emissions resulting from pure coal and pure tyre combustion, Levendis *et al.* (1998b) also studied two different coal/TDF blends (75 wt% coal + 25 wt% TDF; 50 wt% coal + 50 wt% TDF). The emissions of NO_x, SO₂, CO, CO₂, and particulate matter resulting from the combustion of the two blends were in between those of the parent materials, confirming the general trends observed for pure coal and pure tyre (see above). Surprisingly, the PAH emissions from the two fuel blends were only slightly greater than those from pure coal, that is, much lower than expected based on the compositions of the blends. This result possibly suggests that the combustion of the coal in the fuel blends supports the oxidative destruction of PAHs produced during the burning of the tyre particles (Levendis *et al.* 1998b).

In another experiment, de-wired TDF was combusted together with natural gas as the primary fuel in a rotary kiln incinerator simulator (Lemieux 1994). This study examined organic gases, along with emissions of particulate matter and metal aerosols. It revealed that adding TDF to natural gas increased emissions of PAH, particulate matter, and metals, most notably those of Zn, As, Sb, and Pb. Another small-scale study with natural gas and TDF has demonstrated that TDF can be a fuel that is technically viable for reburning, that is, a special method of significantly reducing NO_x emissions (Miller *et al.* 1998).

Besides these laboratory experiments there are several emission studies on controlled tyre co-combustion in large-scale industrial facilities, including cement and lime kilns, utility boilers, and pulp and paper mills (Ohio Air Quality Development Authority 1991; EPRI 1991; EPA

1997; Giugliano *et al.* 1999). Comparison between data sets is often difficult because the combustion conditions were either very different between the various experiments or not reported in detail. Inter-facility comparisons are further complicated by the fact that different units are given for the emission levels. Moreover, most of these studies did not report the composition of the tested fuels. Despite these complications, it is observed that emissions of NO_x generally decreased with increasing amounts of TDF in the fuel blend (e.g., Horvath 1991), whereas emissions of some metals, particularly Zn, increased. In most coal-burning facilities, emissions of particulate matter also augmented with the addition of TDF. Co-firing of TDF with coal resulted in either enhanced or reduced SO₂ emissions, depending on the S content of the coal and TDF used. Overall, the results indicate that properly designed industrial facilities combusting solid fuels can supplement their normal fuels with 10–20 wt% TDF and still comply with environmental regulations for atmospheric emissions (EPA 1997). Dedicated tyre-to-energy facilities, however, are designed to burn exclusively tyres and, due to the high boiler temperatures (about 1400 °C) and sophisticated air pollution control systems, achieve emission rates that are much lower for most compounds than those from other solid fuel combustors (Ohio Air Quality Development Authority 1991; EPA 1997).

Below, we discuss test burns in industrial facilities that combusted both pure coal and coal/TDF blends. These fuels were burned at identical conditions in each combustor: a stoker boiler at the University of Iowa (EPA 1997; Schwarzhoff & Milster, personal communication 2003), a stoker boiler at Purdue University (this study), and two different cement kilns (Carrasco *et al.* 1998; Mukherjee *et al.* 2003).

The University of Iowa conducted test burns with three different fuels (pure coal, and coal/TDF blends of 96/4 wt% and 92/8 wt%), whereas the Purdue University study compared pure coal and a coal/TDF blend of 95/5 wt% (Table 8). In the Iowa test burn results, it is noted that the 96/4 wt% blend did not produce data that are entirely consistent with the trend exhibited by the blend containing 8 wt% TDF. This inconsistency is probably due to heterogeneity within each of the two fuel components or due to insufficient mixing of the two fuels. The Purdue data correspond both qualitatively and quantitatively more closely with the 8 wt% TDF Iowa data than they do with the 4 wt% TDF data (Figs 7 and 8; Table 8). Both the University of Iowa and the Purdue University

Table 8. Comparison of average emissions resulting from combustion of various fuel blends in stoker boilers

	University of Iowa*		University of Iowa*		University of Iowa*		Purdue University		Purdue University	
	100% Coal n = 3	Std. dev.	96% Coal + 4% TDF n = 3	Std. dev.	92% Coal + 8% TDF n = 3	Std. dev.	100% Coal n = 3	Std. dev.	95% Coal + 5% TDF n = 3	Std. dev.
Fuel blend (wt%)	202	1	199	5	199	0.6	183.4	0.7	195	3
Gas temperature (°C)	14	2	10	2	13	1	13.6	0.9	13	2
Total particulate matter (kg/h)	0.04	0.01	0.03	0.01	0.2	0.1	0.20	0.03	0.7	0.2
<i>Metals (g/h)</i>										
Be	9	5	8	1	20	13	60	12	700	169
Mg	—	—	—	—	—	—	11	4	40	10
Al	—	—	—	—	—	—	0.9	0.4	6	1
Ca	—	—	—	—	—	—	0.91	0.00	3.9	0.9
V	0.9	0.2	0.8	0.1	2	2	48.1	—	40	12
Cr	—	—	—	—	—	—	1.1	0.3	4	1
Mn	—	—	—	—	—	—	4.5	0.4	14	3
Co	1.4	0.5	1.8	0.3	4	2	3.3	0.7	14	2
Ni	2.1	0.6	2.5	0.2	9	4	15	2	2,400	741
Cu	40	15	164	9	1600	360	6	2	26	4
Zn	3	2	2.1	0.7	12	2	7	2	11	4
As	—	—	—	—	—	—	0.04	0.00	0.24	0.07
Se	<0.19	—	<0.188	—	0.5	0.1	0.45	—	1.5	0.3
Cd	—	—	—	—	—	—	<0.9	—	1.8	0.4
Sb	—	—	—	—	—	—	0.2	0.1	0.7	0.3
Te	—	—	—	—	3	2	0.45	0.00	0.45	0.00
Ba	1.2	0.6	0.9	0.2	1.2	0.4	0.45	0.00	1.7	0.5
Hg	1.42	0.05	1.2	0.2	—	—	0.45	0.00	20	7
Tl	—	—	—	—	10	4	8	2	—	—
Pb	2.0	0.8	2.3	0.2	—	—	—	—	—	—
<i>Gases (kg/h) †</i>										
CO	5.0	0.4	8	5	6.0	0.2	5	2	3.0	0.5
NO _x	82	9	78	3	70	22	63.2	0.5	67	3
SO ₂	320	16	291	2	290	27	390	22	360	12
HCl	5	1	6.3	0.9	7	3	0.69	0.06	1.1	0.3
Fluoride ‡	0.00080	0.00003	0.00064	0.00007	0.00077	0.00000	0.20	0.01	0.22	0.03
<i>Organics</i>										
THC (mg/s)	265	—	265	—	189	—	120	83	20	23
Dioxins/Furans (ng/s) §	18	4	10	1	6	3	<19	—	<17	—

* Average data for the University of Iowa published by EPA (1997). Data presented here compiled from original University of Iowa data set provided by Schwarzhoff & Milster (personal communication 2003).
† At 7% O₂ by volume.; ‡ Purdue University tested for fluoride as HF. University of Iowa fluoride data not reported at 7% O₂ by volume.; § Purdue University dioxins/furans data reported at 7% O₂ by volume.

ENERGY RECOVERY FROM WASTE TYRES

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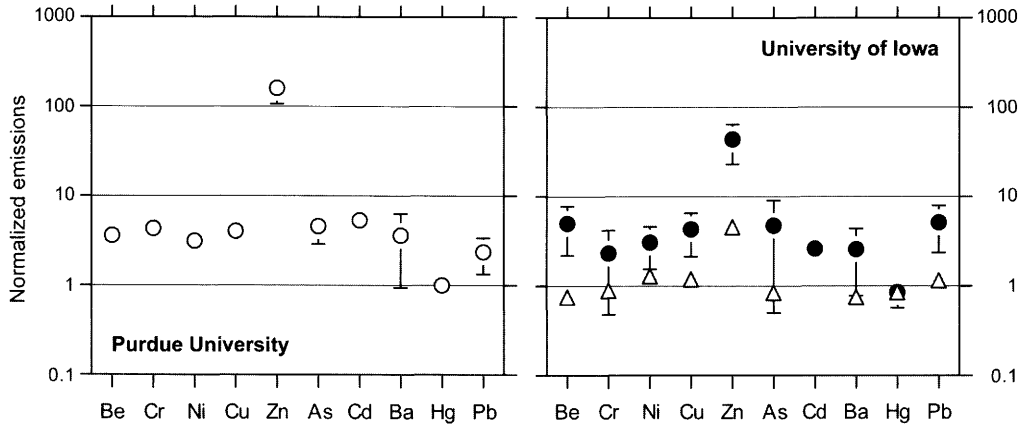


Fig. 7. Comparison of normalized metal emissions from two stoker boilers. Values represent emissions from combustion of coal/tyre blends normalized to the respective emissions from combustion of pure coal. Purdue University data set is for 5 wt% TDF. Symbols for University of Iowa data set: Δ , 4 wt% TDF; \bullet , 8 wt% TDF. Data without error bars have standard deviations that are smaller than the symbols. Data from Table 8.

studies document that combustion of coal/TDF blends with ≥ 5 wt% TDF rather than pure coal leads to significantly enhanced average emissions of Be, Ni, Cu, Zn, As, Cd, and Pb (Fig. 7). Most dramatic are the increases observed for Zn, whose average emission rate was multiplied by a factor of more than 43 when ≥ 5 wt% TDF were added to coal (Fig. 7). In the Purdue test burn, Zn emissions averaged 2.4 kg/h for the 95/5 wt% blend, compared to 15 g/h for the combustion of pure coal (Table 8). The Purdue study further reveals significantly increased emissions in most other analysed metals, including Al, Cr,

and Sb (Table 8). For example, combusting the 95/5 wt% blend led to Al emissions that were 12 times larger than those from combustion of pure coal. On the other hand, emissions of Hg and total particulate matter remained constant in both studies (Table 8; Fig. 7). Total hydrocarbon (THC) emissions decreased in both stoker boiler experiments when the coal + TDF mixture was used as fuel (Fig. 8). The high Cl content (nearly 3000 ppm) of the TDF used in the Purdue test, the only comparative study for which Cl concentrations in the fuel are available, is responsible for the pronounced increase in HCl

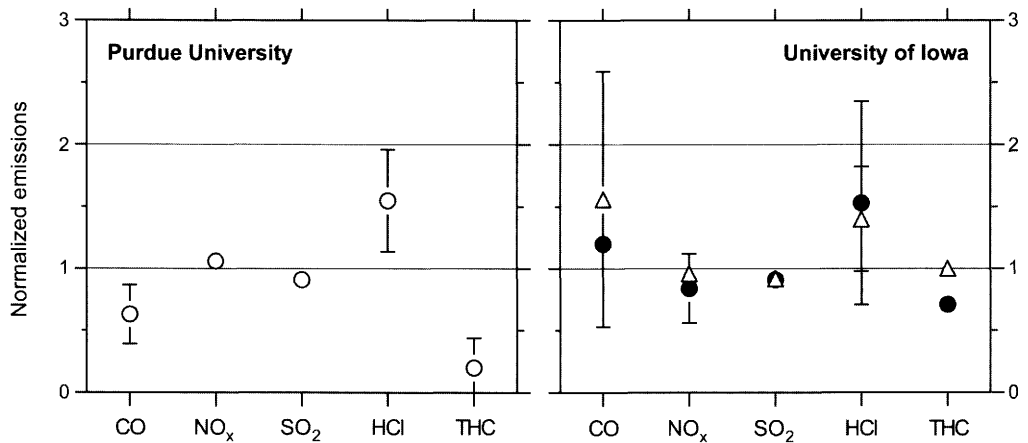


Fig. 8. Comparison of normalized gas emissions from two stoker boilers. Values represent emissions from combustion of coal/tyre blends normalized to the respective emissions from combustion of pure coal. Purdue University data set is for 5 wt% TDF. Symbols for University of Iowa data set: Δ , 4 wt% TDF; \bullet , 8 wt% TDF. Data without error bars have standard deviations that are smaller than the symbols. Data from Table 8.

emissions (Fig. 8) from the blend relative to those from pure coal, which contains only 215 ppm Cl (Table 4). Additionally, the Iowa study showed a notable reduction in dioxin/furan emissions with increasing TDF percentage.

The results reported by Carrasco *et al.* (1998) revealed that nearly all studied metal emissions, measured at the exit of a cement kiln stack, were significantly higher when a blend of 80 wt% coal + 20 wt% TDF was combusted instead of pure coal. Especially notable are increased emissions in Cr, Mn, Cu, Zn, and Pb (Table 9). The exception to this trend is Hg, which exhibited a 30% reduction in its emission rate when the coal + TDF mixture was burned. The data further document reductions in NO_x and organic compound emissions, including PAHs, where the most drastic decrease was observed for dioxins and furans. On the other hand, emissions of CO, SO₂, and HCl increased considerably with the addition of TDF (Table 9). The total particulate emissions from combustion of the blend were only slightly greater than those from pure coal. Carrasco *et al.* (1998) used their data to model atmospheric dispersion of the emitted contaminants in the vicinity of the

cement kiln, and found that the environmental impact was most pronounced within an area with a 500 m radius around the pollution source. Their calculations revealed that the ground-level maximum 24 h concentrations of particulate matter, CO, SO₂, Fe, Zn, Pb, and HCl increased significantly when TDF was added to the fuel, whereas the concentrations of NO_x and several organic compounds decreased. These findings are in agreement with the values measured at the kiln stack exit, but the relative changes are different, because they are influenced also by the direction of the dominant winds and by the presence of obstacles in the flow path (e.g., buildings). It should be noted that Giugliano *et al.* (1999) found in their study of TDF co-firing in a cement kiln that the metal emissions, including those of volatile elements, such as Zn and Pb, were not affected by the addition of TDF to petroleum coke. This somewhat surprising result was attributed to the high particle-removal efficiency of the air pollution control system. On the other hand, Giugliano *et al.* (1999) observed, like Carrasco *et al.* (1998), that addition of TDF led to a considerable reduction in NO_x emissions.

Table 9. Emissions from combustion of fuel blends in a cement kiln*

	Emissions 100% Coal	Emissions 80% Coal + 20% TDF [†]	Normalized emissions [‡]
	<i>n</i> = 1	<i>n</i> = 1	
Gas temperature (°C)	292	273	—
Total particulate matter (mg/m ³)	99.2	106.5	1.07
<i>Metals</i> (µg/m ³)			
Al	745.2	1284.0	1.72
Cr	48.0	315.5	6.57
Mn	27.5	87.3	3.17
Fe	873.0	1862.4	2.13
Cu	3.93	11.2	2.85
Zn	185.1	1700.1	9.18
Hg	55.1	37.7	0.68
Pb	120.8	334.8	2.77
<i>Gases</i> (mg/m ³)			
CO	155.4	228.0	1.47
NO _x	1754.4	1483.9	0.84
SO ₂	696.8	905.2	1.30
HCl	9.68	15.80	1.63
<i>Organics</i> (µg/m ³)			
Chlorobenzene	1.72	0.88	0.51
PAH	83.5	68.4	0.82
Naphthalene	76.3	68.3	0.90
Dioxins/Furans	0.0010	0.0004	0.40

*Data collected at the kiln stack exit (from Carrasco *et al.* 1998).

[†]Percentage is probably by weight, but was not specified in the original study.

[‡]Normalized emissions = Emissions (80% Coal + 20% TDF)/Emissions (100% Coal).

Mukherjee *et al.* (2003) investigated atmospheric emissions during a test burn in a cement kiln in Finland. They compared the heavy metal emissions resulting from combustion of coal/petcoke with those arising from burning a blend consisting of 10 wt% shredded car tyres and 90 wt% coal/petcoke. The diagrams presented by Mukherjee *et al.* (1993) show that adding car tyres led to a significant increase in the total emissions of V, Cr, Mn, Fe, Ni, Cu, Zn, and Pb. These trends are similar to those noted by Carrasco *et al.* (1998). On the other hand, the emissions of Hg and TI decreased considerably as a result of adding tyres to the fuel. Mukherjee *et al.* (2003) further observed that this decrease was accompanied by a shift of Hg and TI from the gaseous phase to particulates and argued that this effect is environmentally beneficial because particulate matter can be controlled more easily. The study of Mukherjee *et al.* (2003), unfortunately, presented mainly diagrams and reported actual emission values for Fe and Hg only; it will therefore not be discussed further (normalized emissions: Fe = 3.12; Hg = 0.11; cf. Carrasco *et al.* 1998 data in Table 9).

Comparing the cement kiln results of Carrasco *et al.* (1998) to the Iowa and Purdue stoker boiler data is difficult because of the differences in types of facilities, compounds analysed, composition of TDF and coal, and percentage of TDF used in the fuel blends. Moreover, emission units of mass per time (Table 8) are less suitable for inter-facility comparison than mass per heat input (e.g., kg/MJ; EPA 1997), which were not available. Nevertheless, there are similar emission trends for a number of compounds analysed in all three test burns: the average metal emissions, when normalized to the emissions from pure coal combustion, are on approximately the same level (Fig. 7, Table 9), despite the fact that the cement kiln used a much higher percentage of TDF in the fuel blend. The exceptions to this trend are Zn and Hg. Zinc appears to be emitted at significantly higher levels when coal + TDF mixtures were burned in the two stoker boilers. This result is difficult to interpret because the Zn contents of the coal and TDF used in the actual experiment are only known for the Purdue test burn (coal: 36 ppm; coal + TDF: 183 ppm; TDF: 10 900 ppm; unpublished data). In the two stoker boiler experiments, the Hg emissions are practically unaffected by the addition of TDF to coal (Fig. 7), whereas a considerable reduction is observed in the cement kiln. Similarities between the three test burns are also observed for the gaseous emissions (Fig. 8, Table 9), but the effect of adding TDF to coal is less pronounced for the gases than for most metals. The reduction in CO emissions displayed

for the Purdue experiment appears to be in contrast to the results from the other studies. On the other hand, the Carrasco *et al.* (1998) data show a marked increase in SO₂ emissions, in contrast to both stoker boiler experiments. This discrepancy probably reflects differences in the S content of the fuel. Reduced SO₂ emissions have also been observed by Giugliano *et al.* (1999) for TDF co-combustion in a cement kiln.

Discussion

The relatively large heating value of tyres makes the use of TDF or whole tyres for combustion in utilities, industry, and manufacturing the most promising method for waste tyre management.

The most obvious advantage to using tyres as a fuel source is the partial alleviation of many of the environmental problems associated with traditional disposal of scrap tyres in stockpiles and landfills. The eradication of tyre piles and landfill waste will reduce many of the risks to human health linked to the buildup of tyre waste. The removal of outdoor tyre stockpiles provides great aesthetic benefits as well.

Tyre combustion not only helps eliminate a waste problem, but also conserves the limited amount of fossil fuel supplies. Because the energy extracted from tyres per mass unit is greater than that of most other fuels, tyre combustion can prove to be a financially beneficial alternative to fossil fuel combustion (e.g., McGowin 1991; Lamarre 1995). In some cases, the use of tyre fuel can decrease the cost of energy production by up to 40% (Ohio Air Quality Development Authority 1991). However, several factors may reduce the economic benefits of using this fuel. The proximity of the tyre source to the facility weighs heavily on the economic benefits garnered by the operating party. As the distance increases, the cost effectiveness of combusting tyres for energy decreases (Jones 1998). It is for this reason that the large tyre-to-energy facility in Modesto, California, is located adjacent to the world's largest tyre dump (Makansi 1992). This measure guarantees a continuous supply of scrap tyres, which is an important requirement for tyre-to-energy facilities. Many newer power plants, however, lie at a reasonably large distance from urban centres, and therefore, at a fairly large distance from major waste tyre sources, thus increasing transportation costs.

In addition to collection and transportation costs, there are major expenses associated with the preparation of the tyres for combustion or with modifications to the fuel feeding systems of power plants (e.g., Goddard 1992; Lamarre

1995). The costs associated with shredding tyres into usable chips increases as the size of the chip decreases. Energy production costs further increase when de-wired tyres are used due to the added expense of wire removal, which is commonly achieved via magnetic or gravity separation (Ohio Air Quality Development Authority 1991; EPA 1994). This financial burden is especially prevalent for facilities using cyclone-fired boilers and for paper and pulp mills, as they typically require small, de-wired tyre chips. Therefore, the most cost-effective process for utilizing tyres as fuel is one that uses whole, unaltered tyres. However, dedicated tyre-to-energy facilities combusting whole tyres are costly to build, even when modifying a pre-existing combustion facility. Other cost drawbacks to tyre combustion are those associated with the installation and maintenance of additional air pollution control devices.

Several studies investigating the environmental effects of controlled tyre combustion have been conducted. It is evident that atmospheric emissions can be greatly reduced if proper air pollution control systems are installed. Laboratory and field data provide evidence indicating that concentrations of some environmental pollutants, especially NO_x, may decrease due to tyre combustion, whereas others increase compared to pure coal combustion. Zinc is an example of an element that increases in both solid combustion products and atmospheric emissions. The geochemical impact of higher Zn contents in fly and bottom ash on leaching processes in disposal sites remains to be tested.

Combustion of tyres as alternate fuel, for example in tyre-to-energy facilities, produces ash with particularly high Zn contents. Such ash might represent a valuable resource for Zn recovery. Makansi (1992) reported that the Stirling tyre-to-energy facility in Connecticut had plans to sell the fly ash to metal refineries for extraction of the Zn. It should be noted that the highest amounts of Zn found in pure tyre ash are 60 wt% (Polasek & Jervis 1994). This concentration is similar to the Zn content of pure sphalerite (ZnS), the most important Zn ore. However, even if the Zn contents of tyre ash are lower (Table 6), recovery of Zn from TDF or tyre ash may be more economical than extraction from sphalerite ore because it does not involve mining for sphalerite and because the bulk ash is a fairly homogeneous and fine-grained material. Moreover, the use of tyre ash as a source of Zn would be an environmentally beneficial process for three reasons: (1) it would not create sulphide waste with its associated environmental problems (e.g., Sidenko

et al. 2001); (2) it would help conserve natural resources; and (3) it would take advantage of a waste material rather than discard it. From an environmental standpoint, specially designed tyre-to-waste facilities therefore represent a promising option for dealing with one of the greatest disposal challenges for solid waste and at the same time contribute to a sustainable energy cycle.

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