Appendix 3

CASE STUDIES: THE MANAGEMENT OF COAL ASH, CO₂ AND MERCURY AS WASTES

CASE STUDIES: THE MANAGEMENT OF COAL ASH, CO_2 AND
MERCURY AS WASTES

This appendix presents case studies on the management of mercury (an example for theme 1 of this study) and coal ash and CO_2 following the development of carbon capture and storage (CCS) (theme 2 of this study).

Two of the primary sources of base load electricity in the future are expected to be coal equipped with carbon capture and storage capability and nuclear energy; both are likely to be need in significant quantities if the world is to meet demanding reductions on emissions of climate change gases. An objective of this study is to examine the differences in the way the waste products from these generation methods are managed. Coal ash and carbon dioxide are the main waste products from combustion of coal to generate electricity and this appendix presents an overview of some of the issues associated with their management. Management of radioactive wastes are considered in detail in Appendix 1. The aim of this appendix is to provide the basis for the broad comparison between the wastes from coal and nuclear electricity production that is presented in Chapter 3.

Mercury is an example of a highly toxic, hazardous metal. This case study explains some of its hazardous characteristics and aims to present a perspective on the management and eventual geological disposal of this highly toxic waste stream. Because the hazard from mercury does not diminish with time, when it is disposed of it must be isolated from man and the environment, effectively forever. In order to cope with safety requirements over long periods, without the need for monitoring and intervention, the trend for managing mercury waste is towards deep disposal. (Brasser, 2009) The long term isolation requirements for mercury wastes are therefore of a similar nature to those for high-level radioactive waste.

A3.1 Coal ash from power production

A3.1.1   *Electricity production share and total production of ash*

In 2005, about 40% of the world's electricity was generated by coal combustion (Couch, 2006), see Figure A3.1-1. Around 3.2 Gt of coal is used worldwide for thermo-chemical energy production each year giving rise to total of up to around 0.6 Gt of ash per year. A typical 500 MWe coal fired power station burns about 2 Mt/a of coal.

There is significant global concern about the climate change effects of CO_2 emissions from fossil fired electricity generation, which dominate anthropogenic releases to the atmosphere. However, other releases also have significant detrimental effects. Air pollution from coal-fired electricity production includes a mixture of pollutants, including fine particulate matter, carbon monoxide, nitrogen dioxide, sulphur dioxide, ozone and volatile organic compounds and inorganic substances. Air pollution control systems in modern coal fired power plants may include a scrubber system where most residues of

sulphur and nitrogen oxides are removed, together with hydrochloric acid. Volatile substances like mercury and cadmium are released, to some extent, into the atmosphere along with fluorine, chlorine and bromine.

A European Environmental Agency study shows that 30% of the total PM_{10} (particles less than 10 microns in diameter) emissions in Europe result from energy production. It states that coal is a significant emitter of PM_{10} during electricity production, and should therefore be considered a significant source of health damage worldwide, even in advance economies. The *OECD Environmental Outlook* estimates that PM_{10} emissions caused 960 000 premature deaths in 2000, with 9.6 million years of life lost worldwide.

Coal combustion also releases radioactivity to the environment. The main sources of radioactivity include uranium, thorium and daughter products such as radium, radon, polonium, bismuth and lead. Although not a decay product, naturally occurring radioactive potassium-40 is also a significant contributor.

Figure A3.1-1: Electricity production share for different fuels in 2005
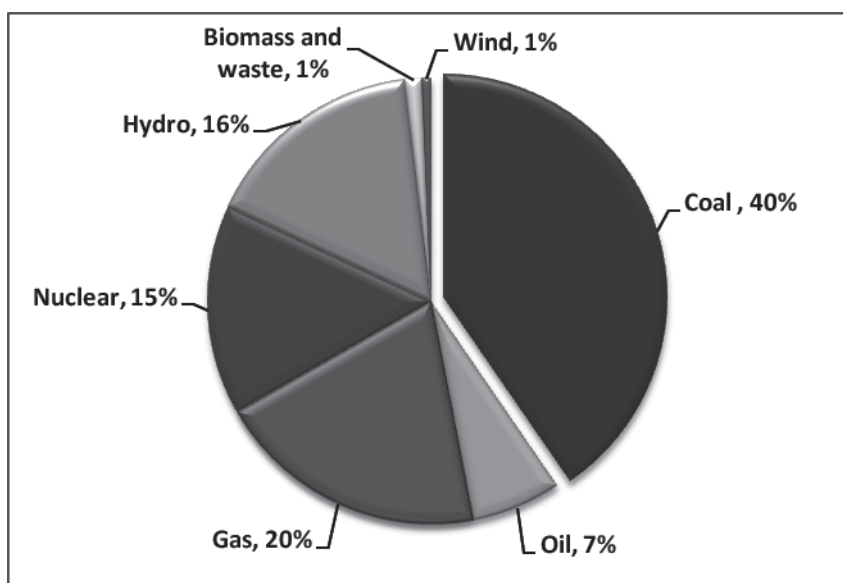
Sources: Couch, 2006; Joshi and Lothis, 1997; Sear, 2001; Sloss, 2007; Barnes and Sear, 2004.

Volumes of coal used and of ash generated for the purpose of power production are given in Table A3.1-1 for coal producing countries and in Table A3.1-2 for non-producing countries. It should be noted that different countries produce these data in different ways that may lead to apparent inconsistencies; these data are taken from a compiled source.

Table A3.1-1: Amounts of coal used and of ash generated in power production for coal producing countries in 2002

Country	Production bituminous coal, Mt/y	Production brown coal, Mt/y	Coal consumption (net import+export) [for coke making]	Used for power production, Mt/y	Ash [Average % in the coal]	Total ash production, Mt/y
China	1 343	50	1 315 (-78) [240]	700-750	[24%]	160-185
United States	520	473	992 (-1) [19]	820	[10-15%]	90
India	335	24	379 (+20) [23]	285-290	[30-40%]	90
Australia	256	84	133 (-207) [4]	55 & 68	[30% & 4%]	18-23
Russia	168	85	214 (-39) [40]	165-175	[10-20%]	25-35
South Africa	215	0	156 (-59) [3]	80-90	[30-35%]	20-30
Germany	29	182	242 (+31) [10]	34 & 169	[10% & 8%]	15-20
Poland	102	85	146 (-41) [14]	40 & 60	[20% & 10%]	10-15
Indonesia	103	0	29 (-74)	20-22	[10%]	2-3
Ukraine	82	1	86 (+3) [27]	40	[20-25]	10-20
Kazakhstan	71	3	50 (-24) [3]	25	[40]	8-15
North Korea	53	15	31	na		na
Greece	0	70	68 (+2)	65	[10-15]	8-12
Canada	30	37	60 (-7) [4]	11 & 44	[10% & 15%]	5-10
Czech Republic	15	48	58 (-5) [5]	4 & 41	[15%]	4-8
Turkey	2	51	65 (+12) [4]	1 & 42	[15-20%]	6-10
Colombia	40	0	3 (-37)	–		–
Serbia & Montenegro	0	34	34	na		na
Romania	4	27	34 (+3) [2]	25	[10-25%]	2-5
United Kingdom	30	0	59 (+29) [5]	46	[10%]	4-6
Bulgaria	0	26	31 (+5) [1]	24	[25-30%]	5-8
Spain	10	12	44 (+22) [4]	37	[10% & 20%]	5-10
Thailand	0	20	20	15	na	na
Vietnam	15	0	10 (-5)	8	na	na
Hungary	0	13	14(+1)[1]	12	na	na
Total	**3 423**	**1 340**	**4 273 [409]**	**≈2 975**		**487-595**

Source: Couch, 2006.

Table A3.1-2: Volumes of coal used and of ashes generated for the purpose of power production for non coal-producing countries in 2002

Country	Coal consumption, Mt/y [production Mt/y]	Used for power production, Mt/y (estimated amounts)	Ash, [Average%] quantity, Mt/y
Japan	160 [1]	85	[12%] 7
South Korea	75 [3]	43	[12%] 5
Taiwan	51	42	[12%] 5
Italy	20 [2]	14	[12%] 1.7
France	19 [2]	9	[12%] 1.1
Brazil	18 [5]	4	[15%] 0.6
Philippines	13 [2]	na	na
Netherlands	13	9	[12%] 1.1
Israel	12	10	[12%] 1.2
Belgium	11	4	[12%] 0.5
Total	**392 [15]**	**≈220**	**≈23 Mt/y**

Source: Couch, 2006.

A3.1.2   *Properties of coal and the combustion process*

Coal is thought to originate from organic matter in the form of peat that has undergone various ageing processes (diagenesis and metamorphosis) during geological times of tens to hundreds of million years. Coal is a sedimentary rock that occurs in layers coalesced and modified from former peat deposits. The orientation is frequently horizontal but many seams are inclined due to folding, faulting and orogenic displacement of the rock.

Coals vary considerably in character. Recent coals having ages less than around 65 million years are often lignites with considerably higher contents of inorganic constituents than the typical value for older coals, which is around 15%. The geological and chemical processes involving high pressures and temperatures, working over time, have compressed and altered plant remains, increasing the percentage of carbon present, and thus producing the different ranks, or varieties, of coal. Coals are classified based on fixed carbon, volatile matter, and heating value. The incombustible matter in coal, which acts to lower the relative amounts of carbon and thus the rank of coal, becomes ash after burning. Minerals represent the inorganic parts of coal and include clay (the most abundant inorganic constituent), carbonates, sulphides and quartz, which were either washed into the original swamp plant materials that ultimately were compressed to form peat, or portions of confining rock beds inadvertently mined with the coal. Radionuclides are incorporated into coal as they may be found in the original peat beds or in layers of interspersed inorganic material, or because of intrusion during or after coalification by leaching from surrounding rocks and soils (EPA, 1973; EPA, 1977; DOI, 1963).

The quality of coals also varies considerably with regard to coking properties. Dry distillation (pyrolysis, heating without access to air) of coal gives rise to gas as well as liquids. The proportions of coke, tar and gas depend highly on the individual type of coal used. The same can be said of the mechanical integrity of the coke that is dependent on formation of tar, which on further heating decomposes to form an efficient binding agent between the grains of the coal.

Coal processing before utilisation and burning in furnaces includes blending, pulverisation, washing and flotation to remove as much incombustible mineral material as possible. This increases the heating content of the coal, and serves to minimise, though not eliminate, the amount of ash and clinker generated in the combustion process. Modern coal-fired thermo-chemical plants utilise pulverised fuel to achieve a good contact between the coal grains and the surrounding gasses. Air jets are used to ensure rapid and efficient contact.

There are two main types of furnaces, those with and those without a fluidised bed of fine sand material. The sand assists in transferring heat from the burning particles and to the heat transfer pipes.

A particle in a coal powder burner oxidises in a few milliseconds (Wooley, *et al.*, 2000). Typical temperatures in the hottest parts approach 1 650°C. The maximum temperature is intentionally kept at least 100°C lower than that for stoichiometric composition in the feed in order to reduce the formation of oxides of nitrogen for which strict limits apply for emission. Additional air is added in the form of jets a little higher in the furnace to ensure excess of oxygen everywhere in the flue gasses. Typical residence time for the fuel particles in the furnace is 3-4 seconds (Wooley, *et al.*, 2000). In this way, the combustion process becomes completed with a high efficiency.

A3.1.3 *Air pollution control systems and means of ash removal*

Some of the ash simply falls down by gravity to the lower parts of the furnace (including the reheater and economiser parts). Other fractions of the ash are collected by means of a cyclone. Frequently, both of these are referred to as "bottom ash" (and they may be mixed in the process of removal) as opposed to the ash leaving the furnace area together with the flue gasses which is referred to as "fly ash". In a modern coal combustion facility, most of the ash (around 80%) (Wooley, *et al.*, 2000) is collected in the form of fly ash.

In the majority of cases, most of the fly ash is removed by electrostatic precipitation.[1] In addition – or alternatively – bag filters[2] may be applied, sometimes in conjunction with dry or semi-dry[3] chemical air pollution control.

Air pollution control in modern coal combustion facilities may also include a scrubber system where residues of sulphur and nitrogen oxides are removed, together with hydrochloric acid. The main reaction product from such systems is gypsum (calcium sulphate).

Mercury, and to a certain extent cadmium, are much more volatile than other heavy metals present in fumes from coal combustion. They do therefore not condense efficiently in the ash and may be emitted and become an environmental and health hazard even if the fumes are cleaned by

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1. The fumes pass areas of high electrostatic fields that make the charged particles move and attach to surfaces from which they are intermittently removed.

2. The ash is removed by recurrent back flushing.

3. Lime sludge is sprayed into the fumes. The feed is adjusted such that the spray will dry before reaching the filters.

mechanical filters (electrostatic filters and bag filters). Efficient removal of these species may be achieved by adding active carbon to the fumes and/or use wet air pollution control systems (scrubber) in which case the combustion residue will be contaminated with material that has not been combusted.

The need for chemical pollution control is strongly dependent on the quality of the coal, mainly its content of sulphur and mercury. It is also strongly dependent on the type of furnace. Fluidised bed types of furnaces have lower temperatures resulting in formation of less nitrogen oxides. The bottom ash from such furnaces, or rather "bed ash" as it is usually called in this case, invariably contains some of the bed material as well.

Thus, no visible combustion fumes leave a modern coal-fired plant. The only "smoke" that can be observed is some condensation in the air leaving the cooling towers (of water previously evaporated inside the tower). Under normal weather conditions, this condensation soon re-evaporates as the air from the cooling towers is mixed with the surrounding air. Large amounts of invisible carbon dioxide leave the stacks from coal-fired thermo-chemical plants, and this is of great concern since it is a major contributor to climate change.

The means of removing ash vary, and may not be in exact one-to-one correspondence to the processes installed. It was noted above that bag filters for fly ash might be combined with semi-dry chemical air pollution control. The ash removal systems may be designed in such a way that ash from individual removal points may not be taken out separately. This may apply to ash from different units as well.

The bottom ashes (and/or bed ashes and/or cyclone ashes) in particular may be very hot at the point where they are removed. This may make them difficult to handle due to the continuing combustion of residues of burnable material. Therefore, such ashes are often removed by passing them through a water bath. Wet ashes are handled and managed separately from dry ones.

A3.1.4   *Ash classification schemes*

Not unexpectedly, ashes are classified differently in different countries and also between different utilities, plants and combustion units. Interpretations and usage of the various terms may vary since precise definitions of the categories may exist only at a particular plant level. It is important to note that the categories include large-volume categories as well as small-volume ones, and that the pertinent strategies for the management of the ashes may vary considerably depending on the volume of the material in question.

As an example, the United States Environmental Protection Agency in their *Report to Congress on Wastes from the Combustion of Fossil Fuels* [EPA, 1999] uses the following categories for the large volume residues:

- fly ash;
- bottom ash;
- boiler slag;
- flue gas desulphurisation (FGD) sludge.

The following categorisation is mentioned for small-volume residues:

- coal pile runoff;
- coal mill rejects/pyrites;
- boiler blow-down;
- cooling tower blow-down and sludge;

- water treatment sludge;
- regeneration waste streams;
- air heater and precipitator wash water;
- boiler chemical cleaning waste;
- floor and yard drains and sumps;
- laboratory wastes;
- waste water treatment sludge.

It should be remembered that even with a perfect categorisation, there are variations in properties from one time to another. The main reasons for this are variations in the fuel and in the thermal load of the unit in question. In addition, since the residues are usually reactive with regard to moisture and carbon dioxide, the properties may vary with time after the waste products have been removed.

Some reasons were given in the previous section why ashes from different removal points in the same unit usually have very different properties. However, the most important differences are those related to chemical composition and to the partitioning processes that take place because of evaporation and fractional condensation in the furnace. A contributing factor here is also differences in thermal history, e.g. differences in the rate of cooling.[4]

It is the larger ash particles that form the bottom ash while fly ash has a small particle size where most of the material is in the 0.005-0.02 mm range. (Wooley, *et al.*, 2000) The reason for the small particle size and the partitioning with regard to particle size is the transient and rapid events in the furnace. There is little time for diffusion of condensing matter to the larger particles, and therefore volatile material preferentially condenses on the small particles.

A classification of a number of trace elements in coal ash with respect to their behaviour in a furnace environment is presented in Table A3.1-3.

Table A3.1-3: Classification of trace elements with regard to their volatility in a furnace environment

Group	Elements
3	Hg, Br, Cl, F
2+3	B, Se, I
2	As, Cd, Ga, Ge, Pb, Sb, Sn, Te, Tl, Zn
1+2	Ba, Be, Bi, Co, Cr, Cs, Cu, Mo, Ni, Sr, Ta, U, V, W
1	Eu, Hf, La, Mn, Rb, Sc, Sm, Th, Zr

Note: The elements in Group 3 are the most volatile, and those in Group 1 are the least volatile.

Source: Sloss, 2007.

The major elements are not included in Table A3.1-3. They are nonetheless important since there is a competition between various elements with regard to e.g. chlorine. Thus, sodium and potassium are over-represented in the fly ash. They tend to condensate as chlorides. Silicon and aluminium are over-represented in the bottom ash while calcium and magnesium may not exhibit a preference.

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4. Very rapid cooling (quenching) gives rise to a more reactive material as compared to slow cooling (other factors being the equal).

A3.1.5 *General chemical composition of coal and coal combustion residues*

The chemical composition of coal and the major elements[5] in the corresponding ash is presented in the form of a few examples in Table A3.1-4b, c. The examples are taken from a wide range of coals, mainly from exporting countries. The work was carried out at a test facility; therefore, the ash in this case represents all of the ash except that which is typically absorbed in the chemical cleaning of the flue gasses. Trace elements in coal and their intervals of occurrence are presented in Table A3.1-4a.

Table A3.1-4a: Trace elements in international thermal coals compared with Australian coals, (mg/kg)

Element	International coals			Australian coals		
	Average	Low	High	Average	Low	High
As	3.3	0.32	26	0.93	0.1	2.7
B	59	6	143	21	4	36
Be	0.95	0.1	3.2	0.82	0.2	2.1
Br	7	2	38	5	2	17
Cd	0.07	0.01	0.19	0.09	0.01	0.28
Cl	310	10	1 470	320	10	1500
Co	4.7	1	13	3.7	1.2	12
Cr	12	2	34	9	2.9	24
Cu	9	1	28	14	6.2	32
F	100	15	305	98	35	340
Hg	0.066	0.01	0.19	0.021	0.006	0.08
I	3	2	7	6	2	14
Mn	44	8	123	99	4	700
Mo	1.1	0.07	4.2	0.85	0.1	2.7
Ni	9	1	22	8.6	1.4	31
Pb	7.2	0.5	22	5.8	2.2	14
S.%	0.65	0.115	3.0	0.6	0.21	0.95
Sb	0.37	0.02	1.4	0.46	0.05	1.2
Se	1.4	0.1	5.3	0.47	0.12	1.1
Th	3.1	0.1	12.2	2.6	0.5	6.9
U	1.2	0.02	5.5	0.93	0.27	2.5
V	20	1.5	54	23	7	62
Zn	12	4	55	14	4	51

Sources: Couch, 2006; Dale, 2005.

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5. The major elements are figured as hypothetical formula units.

Table A3.1-4b: Examples of the chemical composition of coal and the corresponding ash (continuation)

Coal	Harworth	Bailey	PRB1	PRB2	Bowen basin	Hunter Valley	Prodeco	Goedehoop	Talcher	JR/PRB1 blend	PRB1/Baley blend
Origin	UK	Eastern US	Western US	Western US	Australia	Australia	Colombia	South Africa	India	US	US
H_2O, %	2.5	2.3	18.0	19.7	4.8	3.2	3.7	2.8	9.7	18.9	11.2
ash, %(ar)	14.4	8.9	3.7	5.8	7.6	9.8	8.6	13.1	39.7	4.8	5.6
VM, %(ar)	31.4	34.5	34.5	35.0	27.8	30.7	35.5	25.4	24.0	43.7	34.8
GCV, MJ/kg(ar)	28.9	31.0	24.1	22.5	28.9	29.9	29.6	28.2	15.1	23.4	27.5
NCV, MJ/kg(ar)	27.8	29.9	22.9	21.3	28.0	28.9	28.5	27.4	14.3	22.2	26.3
S, %(ar)	2.3	1.3	0.33	0.33	0.4	0.46	0.66	0.7	0.37	0.46	0.8
Cl, %(ar)	0.20	0.21	0.01	0.01	0.01	0.02	0.5	0.01	0.01	0.02	0.17
C, %(daf)	82.4	83.7	76.3	77.4	81.8	83.0	81.4	83.5	74.5	75.9	80.6
H, %(daf)	5.5	5.3	4.4	4.4	4.5	4.9	5.5	4.5	4.9	4.4	4.9
N, %(daf)	1.78	1.66	0.98	1.12	1.77	1.86	1.68	2.03	2.07	1.1	1.41
O, %(daf)	7.4	7.7	17.9	18.6	11.5	9.7	10.1	9.2	17.7	18.0	11.9
VM, %(daf)	37.8	38.9	44.1	47.0	31.7	35.3	40.5	30.2	47.4	45.2	41.8
Fuel ratio	1.65	1.57	1.27	1.13	2.15	1.83	1.47	2.31	1.11	1.20	1.39

Notes: Coal analysis determines the amount of fixed carbon, volatile matters (VM), moisture and ash within the coal sample. The variables are measured in weight percent (wt. %) and are calculated in several different bases. AR (as-received) basis is the most widely used basis in industrial applications. AR basis puts all variables into consideration and uses the total weight as the basis of measurement. DAF (dry, ash free) basis neglect all moisture and ash constituent in the coal. GCV is gross calorific value; NCV is net calorific value.

Sources: Couch, 2006; Wigley and Williamson, 2005.

Table A3.1-4c: Normalised ash compositions, (wt%) (continuation)

Coal	Harworth	Bailey	PRB1	PRB2	Bowen basin	Hunter Valley	Prodeco	Goedehoop	Talcher	JR/PRB1 blend	PRB1/Bailey blend
Origin	UK	Eastern US	Western US	Western US	Australia	Australia	Colombia	South Africa	India	US	US
SiO$_2$	50.8	56.4	36.3	39.2	61.5	81.6	63.4	43.1	67.2	38.6	48.5
Al$_2$O$_3$	26.1	25.4	19.7	20.9	31.0	13.2	20.0	33.3	24.3	19.1	24.9
Fe$_2$O$_3$	14.5	10.7	6.2	6.8	4.1	2.8	7.2	4.8	2.9	6.4	9.7
CaO	1.2	2.1	20.9	23.0	0.5	0.3	2.5	10.9	1.1	22.6	7.6
MgO	1.2	0.9	5.6	4.6	0.2	0.3	2.4	2.6	0.8	5.4	2.4
K$_2$O	3.9	2.3	0.7	0.5	0.3	0.9	2.4	0.5	1.8	0.5	2.0
Na$_2$O	0.8	0.5	7.6	1.5	0.1	0.1	0.8	0.3	0.1	4.0	3.0
TiO$_2$	1.0	1.6	1.8	1.8	1.9	0.7	1.0	1.7	1.3	1.8	1.5
BaO	0.1	0.1	0.9	0.7	0.1	0.03	0.2	0.4	0.1	0.8	0.3
Mn$_3$O$_4$	0.05	0.03	0.06	0.03	0.01	0.03	0.07	0.08	0.03	0.05	0.02
P$_2$O$_5$	0.3	0.15	0.21	1.1	0.39	0.07	0.16	2.4	0.46	0.68	0.16
SiO$_2$/Al$_2$O$_3$	1.94	2.22	1.84	1.87	1.99	0.20	3.18	1.29	2.76	2.02	1.95
Base/acid ratio	0.26	0.20	0.71	0.59	0.05	0.05	0.18	0.24	0.07	0.66	0.33

Source: EPA, 2006.

A3.1.6   Environmental and health properties of coal ash

The environmental and health properties of coal ash are determined by examining the exposure pathways. Generally, oral intake of liquids (drinking water) and solids (including food) together with inhalation are the pathways considered for exposure for most hazardous substances. In most cases, oral intake from drinking water is the dominant exposure pathway for inorganic components and organic compounds to humans.

For radioactive elements arising from coal combustion, external radiation and inhalation need to be considered as well. For radioactive elements, the principal exposure pathways are through external radiation and inhalation (radon gas and particulates), but this varies by radionuclide and radiation source of exposure.

Use of efficient particle filters at thermo-chemical coal-fired plants has reduced inhalation impacts from smoke stack emissions, but not necessarily in other exposure situations.

Exposure scenarios for living organisms other than humans may be dominated by uptake from surface and groundwater as well as direct radiation exposure. However, these protection criteria are currently designed for protection of populations, not individuals primarily due to lack of data and understanding of health and environmental impacts to animal systems.

Inorganic compounds

Typical leach data for shake tests[6] can be found in Table A3.1-5. The test used resembles the European Union standard test prEN 12457-2 for acceptance for landfills.

Table A3.1-5: Typical ranges for leach data (in mg/litre) for ashes from the United Kingdom using the shake test DIN 38414-S4

Element	Typical range of leachable elements	Element	Typical range of leachable elements
Aluminium	<0.l*-9.8	Magnesium	<0.l*-3.9
Arsenic	<0.l*	Manganese	<0.l*
Boron	<0.l*-6	Molybdenum	<0.l*-0.6
Barium	0.2-0.4	Sodium	12-33
Calcium	15-216	Nickel	<0.l*
Cadmium	<0.l*	Phosphorus	<0.l*-0.4
Chloride	1.6-17.5	Lead	<0.2*
Cobalt	<0.l*	Sulphur	24-510
Chromium	<0.l*	Antimony	<0.01*
Chromium VI	<0.l*-l	Selenium	<0.01*-0.15
Copper	<0.l*	Silicon	0.5-1.5
Cyanide	<0.01*	Tin	<0.l*
Fluoride	0.2-2.3	Titanium	<0.l*
Iron	<0.l*	Vanadium	<0.l*-0.5
Mercury	<0.01*	Zinc	<0.l*
Potassium	1-19	pH	7-11.7

*   Value below detection limit. Water to solids ratio is 10/1 litres per kilogram. The data include a seawater-conditioned sample; hence, the high chloride values.

Source: Sear, 2001.

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6.   Where a sample is gently shaken or tumbled for 24 hours with e.g. ten times its dry weight of de-ionised water.

Organic compounds

The presence of organic compounds such as polycyclic aromatic hydrocarbons (PAH) and dioxin are of constant concern. Historically, their impact on health has been huge due to bad combustion and lack of air pollution control (APC). Extensive research has been carried out to reduce these emissions. Today emissions and their impact are low due to relatively extensive efforts at power plants (the APC building is usually much larger than the furnace building). However, it is difficult to extract all PAH and dioxin from the ash and so there is a debate as to whether it is all measured.

Two classes of organic compounds are of primary interest from a health and environment point of view: polyaromatic hydrocarbons and dioxins. Each of these classes comprises a number of different individual compounds of variable toxicity. Some of the species are very toxic, and may also be carcinogenic, and consequently they have to be restricted to very low levels. Even though the content of polyaromatic carbons in ash is low, the volumes of coal combusted are large.

Extensive research has been performed to evaluate the levels of these compounds in ashes from power production. According to a review in 1995 (Sear, 2001; Wild and Jones, 1995) the major source in the environment, apart from gasworks sites, were found to be coal-fired electricity generation (3 140 tonnes per year in the United Kingdom). These results have been challenged to some extent. It has been said that polyaromatic hydrocarbons in ash are not available to the environment (the half-life of dioxin in ordinary soil is about 2 years), and leach tests in accordance with the method of the United Kingdom Environment Agency have indicated levels for the major species to be less than 0.2 micrograms per litre.

According to Sear (2001), dioxins are unlikely to form under conditions found in coal combustion furnaces, and only traces can be expected in the resulting ash. Various researchers (Sear, 2001) have confirmed that no dioxins over 0.000025 mg/kg are generally found in ashes from coal-fired power plants. This is similar to levels found in typical soils. However, more recent research (Sear, Weatherley and Dawson, 2003) with reference to (JEP, 2003) reports that more efficient techniques have been utilised to extract the polyaromatic carbons from the ash resulting in total values up to 25 mg/kg, though more than half of the values determined were reported to be less than 10 mg/kg. Even if the new data represents significantly higher values than those reported previously, the overall values are relatively low.

Radioactive elements

All of the radon present in the coal is emitted to the air during combustion. (Smith, *et al.* 2001) However, the source for future generation of radon remains in the coal ash. Radon has three radioactive isotopes (see Table A3.1-6).

Table A3.1-6: The isotopes of radon

Natural decay series	Isotope	Named as	Half-life
Uranium	^{222}Rn	Radon	3.82 days
Thorium	^{220}Rn	Thoron	55 seconds
Actinium	^{219}Rn	Actinon	4 seconds

Source: Brune, *et al.,* 2001.

It is clear from the half-lives shown in Table A3.1-6 that radon gases formed in the ash will reach near equilibrium with their parents in periods of between one minute and two months. The radon

present in the coal at the time of combustion leaves via the stack during combustion and so does not appear in the ash. However, this is the case only for a short time as the radon gases then "grow back" into the ash. It is important to be aware that radon behaves differently from all other potentially hazardous components.

The source for Radon-222 is Radium-226, which has a half-life of 1 620 years. The chemistry of radium is very similar to that of barium, which probably acts as a carrier for the radium. According to Chandler, *et al.* (1997), barium is not emitted with the flue gasses but stays in the ash, who states that the radioactivity stays in the ash on combustion (with the exception of the radon already formed).

Actual data on radionuclide content of various coal ashes can be found in Table A3.1-7 and data on natural radionuclide in building materials and extract of relevant parts are presented in Table A3.1-8.

Table A3.1-7: Radioactivity in some coal fly ashes (Bq/kg)

Reports from	Ash from	U-Series			Th-series		
		Min	Max	Average	Min	Max	Average
Germany	Germany	93	137	119	96	155	121
	United Kingdom	72	105	89	3	94	68
	Australia	7	160	90	7	290	150
	Poland			350			150
				189			118
Italy	Italy	130	210	170	100	190	140
Denmark	Denmark	120	210	160	66	190	120
Sweden	Sweden	150	200		150	200	
Belgium	Belgium	112	316	181	88	277	150
Spain	Spain	80	106	91	77	104	89
Czech Republic	Czech Republic	35	190	129	62	142	90

Sources: UNIPEDE/EURELECTRIC, 1997; EPA, 1995; EPA, 1984; Push, *et al*, 1997; IAEA, 2003.

Table A3.1-8: Extract of data for concrete and coal ash from European Commission report

Material	Typical activity concentration (Bq/kg)			Maximum activity concentration (Bq/kg)		
Building material	Ra-226	Th-232	K-40	Ra-226	Th-232	K-40
Concrete	40	30	400	240	190	1 600
Coal fly ash	180	100	650	1 100	300	1 500

Source: EC, 1999a.[7]

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7. According to the foreword, a working party of the Group of Experts established under the terms of Article 31 of the Euratom Treaty has examined the issue of regulatory control of building materials with regard to their content of naturally occurring radionuclides.

The working party developed guidance based on a study providing information about natural radioactivity in building materials and relevant regulations in Member States. This guidance was adopted by the Article 31 Group of Experts at its meeting on 7-8 June 1999 and was published with a view to harmonisation of controls by Member States, in particular in order to allow movement of building products within the European Union.

This guidance was expected to be a useful reference document for the European Commission when considering possible regulatory initiatives at Community level. The Member States have now implemented the Euratom Directive in their national legislation, but despite the Commission's guidance documents, there may very well be significant differences in the national regulations. (Van der Steen, 2006)

Typical concentrations are population-weighed national means of different Member States. Maximum concentrations are maximum values reported in EC (1999b). Higher values might have been reported elsewhere.

A3.1.7 Recycling of coal ash versus disposal

Fraction of coal ash that is recycled

The fraction of coal ash that is recycled varies significantly between countries. Some country specific data can be seen in Table A3.1-9 (United States), Table A3.1-10 (15 EU countries), Table A3.1-11 (Canada) and Table A3.1-12 (Japan). The structure of these tables differs to reflect the different structuring of combustion categories in these countries.

Table A3.1-9: Generation of various residues in 2002 from coal-fired power plants in the United States together with their utilisation (units: tonnes)

Category of residue	Total generation	Total utilisation	Utilisation %
Fly ash	76 500 000	26 628 881	34.8
Bottom ash	19 800 000	7 689 589	38.8
Gypsum*	11 400 000	7 770 000	68.2
Wet scrubbers*	16 900 000	560	3.3
Boiler slag	1 919 579	1 549 972	80.8
Dry scrubbers*	935 394	371 404	39.7
Other*	0	0	
Fluidised bed combustion ash	1 248 599	95 341	76.4
Total	128 703 572	45 523 256	35.37

*   From desulphurisation.

Source: Barnes and Sear, 2004. Data from plants responding to survey extrapolated to include all except for categories in italics for which no extrapolation was carried out.

Table A3.1.10: Generation of various residues in 2002 from coal-fired power plants in Europe (EU 15*) together with their utilisation
(units Mtonnes)

	Ash production	Total utilisation excluding reclamation	Total utilisation excluding reclamation %	Total utilisation including reclamation	Total utilisation including reclamation %	Reuse of stockpiled coal combustion residues	Total production
Fly ash	39.947	18.745	46	35.755	88	0.638	40.585
Bottom ash	5.84	2.42	41	5.211	89	0	5.84
Boiler slag	2.24	2.24	100	2.24	100	0	2.24
Fluidised bed-ash	1.06	0.568	54	0.711	67	0	1.06
Other	0.218	0.218	100	0.218	100	0	0.218
Spray dry absorption-product	0.515	0.297	58	0.482	94	0	0.515
Flue gas desulphurisation-gypsum	9.767	7.088	73	8.326	85	0	9.767
Total	59.587	31.576	52.4	52.943	87.9	0.638	60.225

* EU 15 = Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxemburg, Netherlands, Portugal, Spain, Sweden and United Kingdom.

Source: Barnes and Sear, 2004.

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Table A3.1-11: Generation of various residues in 2002 from coal-fired power plants in Canada together with their utilisation (units: Mt)

Category of residue	Total generation	Disposed/stored	Removed from storage	Total use	Utilisation %
Fly ash	5.030	3.985	0	1.094	22
Bottom ash	1.558	1.472	0.138	0.196	13
Gypsum*	0.421	0	0	0.570	135
Other	0.128	0.124	0	0	0
Total	**7.137**	**5.582**	**0.138**	**1.860**	**26.1**

*   From desulphurisation.

Source: Barnes and Sear, 2004.

Table A 3.1-12: Coal consumption for energy production and generation of coal ash together with the degree of utilisation in Japan during 2001-2005 (units: Mt)

Fiscal year	Coal consumption	Total ash generation	ash content %	Utilisation	Utilisation %
2001	59.159	6.785	11.5	5.271	77.7
2002	64.251	6.920	10.8	5.495	79.4
2003	68.981	7.475	10.8	6.105	81.7
2004	74.270	8.052	10.8	7.128	88.5
2005	78.092	8.334	10.7	7.899	94.8

Source: Watanabe, personal communication.

Specific uses of coal ash in society

The overall prerequisites for use and disposal of residues from coal combustion are that the practice should be:

1.   sound and acceptable from a health and environment perspective;

2.   technically feasible;

3.   logistically feasible.

Although the levels and availabilities of various potentially harmful species are low or moderate in coal combustion residues, it is important that each case be evaluated based on its specific conditions. The presence of certain species at elevated levels may prohibit or impede utilisation for certain purposes, e.g. as soil amendment.

The technical feasibilities include a number of possible properties:

1.   Fineness, such that voids can be filled and reactivity is high.

2.   Rounded shape of the (fly ash) particles such that such that the shear resistance is low (good flow properties) in slurries with high particle loadings. This facilitates mixing, filling up of pore space, compacting, etc.

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3. Pozzolanic[8] reactions (fly ash) improve properties of concrete and mortar above that of good pore filler. It makes the material tighter to penetration of water and more resistant to chemicals and weathering.

4. Low heat of curing (fly ash) facilitates the use in large constructions.

5. Good draining properties make a material (e.g. bottom ash or bed sand) useful in geotechnical constructions.

6. Content of fertilisers and alkaline buffer capacity are valuable in additives to soil.

Data on the specific uses of coal combustion residues in the United States is presented in Table A3.1-13. The data in Table A3.1-13 correspond to the data in Table A3.1-9. Data on the various specific uses of coal combustion residues in 15 countries in the European Union are presented in Tables A3.1-14. The data in Table A3.1-14 correspond to the data in Table A3.1-10.

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8. Some activated silicate-aluminate systems react with lime. They are called pozzolana after the Pozzol volcano where the Romans found material for their cement. It was made of a mixture of lime and volcano ash or a mixture of lime and crushed burnt clay.

Table A3.1-13: Generation of various residues in 2002 from coal-fired power plants in the United States together with their utilisation (units: tonnes)

	Coal combustion residue category =>	Fly ash	Bottom ash	FGD gypsum*	Wet scrubbers*	Boiler slag	Dry scrubbers*	FGD other*	FBC ash
1.	Concrete/concrete products/grout	12 579 136	406 255	60 606	0	9 000	35 436	0	0
2.	Cement/raw feed for clinker	1 917 690	585 480	303 807	0	0	3	0	0
3.	Flowable fill	455 018	0	0	0	0	1 014	0	0
4.	Structural fills/embankments	4 200 982	2 046 545		427 000	12 103	0	0	0
5.	Road base/sub-base/pavement	767 182	1 472 291		616	4 484	2 558	0	0
6.	Soil modification/stabilisation	904 745	98 509	0	0	0	0	0	0
7.	Mineral filler in asphalt	103 173	96 218	0	0	38 496	2 852	0	0
8.	Snow and ice control	2 645	767 455	0	0	8 612	0	0	0
9.	Blasting grit/roofing granules	61 964	137 455	0	0	1 440 706	0	0	0
10.	Mining applications	1 888 855	802 582	0	131 600	0	258 043	0	760 000
11.	Wallboard	0	0	7 247 856	0	0	0	0	
12.	Waste stabilisation/solidification	3 187 773	19 091	0	0	0	67 053	0	193 410
13.	Agriculture	0	6 873	77 700	0	0	0	0	
14.	Aggregate	0	678 109	6 216	0	3 200	1 448	0	
15.	Miscellaneous/other	559 718	572 727	73 815	784	33 371	0	0	
	CCP Category use totals	26 628 881	7 689 589	7 770 000	560 000	1 549 972	371 404	0	953 410
	CCP Category production totals	76 500 000	19 800 000	11 400 000	16 900 000	1 919 579	935 394	0	1 248 599

* From desulphurisation. FDG = flue gas desulphurisation. FCB = fluidised bed combustion.

Source: Barnes and Sear, 2004. Data from plants responding to survey extrapolated to include all except for categories in italics for which no extrapolation was carried out.

Table A3.1-14: Generation of various residues in 2002 from coal-fired power plants in Europe (EU 15*) together with their utilisation (units Mt)

Ash utilisation (Mt)		Fly ash	Bottom ash	Boiler slag	FBC ash	Other	SDA-product	FGD-gypsum	Total	%
Cement raw material	1	4 465	170						4 635	7,7
Blended cement	2	2 042	122		16				2 180	3,6
Concrete addition	3	5 510	0	150	4				5 664	9,4
Aerated concrete blocks	4	746	16						762	1,3
Non-aerated concrete blocks	5	342	1 169						1 511	2,5
Lightweight aggregate	6	107	0				2		109	0,2
Bricks + ceramics	7	90	27			18			135	0,2
Grouting	8	523		170	3				696	1,2
Asphalt filler	9	187							187	0,3
Subgrade stabilisation	10	188	81		41				310	0,5
Pavement base course	11	356	195	1 220	55				1 826	3
General engineering fill	12	1 589	474		4	67	78		2 212	3,7
Structural fill	13	1 445	119		52				1 616	2,7
Soil amendment	14	94	13				0		107	0,2
Infill	15	616	11		368		147		1 142	1,9
Blasting grit	16	22		580					602	1
Plant nutrition	17	4				22			26	0
Set retarder for cement	18							760	760	1,3
Projection plaster	19							726	726	1,2
Plaster boards	20							4 131	4 131	6,9
Gypsum blocks	21							226	226	0,4
Self levelling floor screeds	22							1 239	1 239	2,1
Other uses	23	419	23	120	25	133	48	6	774	1,3
Total utilisation 1-23		18 745	2,42	2,24	568	218	297	7 088	31 576	52,4

* Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxemburg, Netherlands, Portugal, Spain, Sweden and United Kingdom. SDA = spray dry absorption. FGD = flue gas desulphurisation.

Source: Barnes and Sear, 2004.

A3.1.8   *Waste acceptance and disposal*

In the European Union, there are three types of landfills: for inert waste, for non-hazardous waste and for hazardous waste. The acceptance of waste is dictated by the Council Decision of 19 December 2002 establishing criteria and procedures for the acceptance of waste at landfills. (EC, 2003) This decision is implemented in the legislation of the various member countries.

According to the acceptance criteria, a number of specific waste categories are mentioned together with the respective destinations allowed. Residues from coal combustion are not included in these listings.

In general, wastes not specifically listed are to undergo so-called "basic characterisation" which implies short-term shake and column tests. The values obtained in these tests are compared with limits listed in tables for landfills for inert, non-hazardous and hazardous waste. Waste that does not meet the criteria even for acceptance at a landfill for hazardous waste cannot be deposited, but has to be treated until it meets any of the criteria.

There is one exception to this, and the following is stated in section 2.2.1:

"Municipal waste as defined in Article 2(b) of the Landfill Directive that is classified as non-hazardous in Chapter 20 of the European waste list, separately collected non-hazardous fractions of household wastes and the same non-hazardous materials from other origins can be admitted without testing at landfills for non-hazardous waste."

Consequently, in Europe, residues from combustion of coal may be deposited on landfills for inert, non-hazardous or hazardous waste depending on their chemical compositions as well as on their leaching properties.

The broad waste management strategy is similar in the United States. Generally, non-hazardous waste can be deposited on ordinary landfills, and hazardous waste can be deposited at landfills for hazardous waste if the leach criteria are met. In the United States, residues from combustion of coal have been classified as non-hazardous by the Environment Protection Agency (EPA)[9] and this classification has been adopted in many states. Sates have the right to impose their own, more demanding classification and some have established testing conditions (including leach tests) or landfill design requirements for disposal. There have been instances where naturally occurring radionuclides have posed an environmental problem. The state of New Jersey does not allow fly ash to be used as daily cover because of its radioactivity. (NJUS, 2009) In two cases, landfilled coal ash has contributed to the radon and radionuclide levels of Superfund sites. (EPA, 1996, 2005b)

Of concern is that conditions of extreme pH in groundwater are common in ash disposal areas associated with coal-fired power plants. (NRC, 1984) This relationship of pH to uranium leaching is important because uranium is soluble in both alkaline and acidic conditions. Radium, to a smaller extent, is also soluble in water and both uranium and radium may be found in coal ash. A discussion of this matter is found in EPA (2007) which references associated publications on leachability of radionuclides.

In the most cases however, in Europe as well as in the United States, residues from coal combustion may be expected to pass the criteria for disposal on sites for non-hazardous waste.

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9.   There is apparently now some reconsideration of this classification.

A3.1.9 *Coal ash from power production – summary*

- Around 40% of the world's electricity is generated using around 3.2 Gt/a of coal and creating 0.5 to 0.6 Gt/a of ash. The mass of these ash residues are 13 to 16% of the initial coal mass.

- In most countries coal ash is not regarded as a hazardous waste.

- Table A3.1-15 provides a perspective on the global quantities of selected elements that are released to the environment primarily in gaseous form or primarily as ash. These data assume elemental concentrations in international coal (see Table A3.1-4) and a combustion rate of 3.2Gt/a.

Table A3.1-15: Global discharge rates of some elements from coal generation plants

Examples of elements released primarily in gaseous form	Global discharge rate (t/a)
Mercury	210
Bromine	22 000
Fluorine	320 000
Chlorine	990 000
Examples of elements released primarily with ash	
Beryllium	3 000
Uranium	3 800
Thorium	9 900
Arsenic	11 000
Lead	23 000

- In the United States, about 35% of coal ash is recycled (46 Mt/a) whilst in the former EU15 about 88% is recycled (53 Mt/a).

- Coal ash generally has low specific radioactivity, with average concentrations ranging from 157 Bq/kg in the United Kingdom to 500 Bq/kg in Poland. Maximum radioactivity concentrations of 2 900 Bq/kg have been reported.

- The main recycling uses of coal ash are:

  - concrete products and cement;

  - structural fills and embankments;

  - road base construction;

  - mining applications.

- In addition, calcium sulphate produced from flue gas desulphurisation plants is recycled into wallboards and boiler slag is reused for grit blasting.

Clearly, the world of coal ash is different to that of radioactive waste in many respects, for example:

- In comparison with radioactive waste, the solid residues from coal generation have very large mass.

- A large fraction of the residue is reused in the economic cycle to replace large volumes of virgin raw materials; very little radioactive waste is recycled.

- Because such a large fraction of coal residue is reused, the distinction between a waste and a product is not as clear-cut as it is for radioactive waste.
- The nature and oversight of the regulations as well as the waste acceptance criteria for waste disposal are less demanding for coal residues.
- However, the ethical principles that form policies for the management of the two waste types, including the overall aim to protect the environment, are broadly similar.

A3.2 Mercury containing waste

A3.2.1 Background

Because of its unique chemical and physical properties, mercury has proved to be useful in numerous products and chemical processes. As a result, mercury is present throughout the environment and levels have increased over time. Because of its toxicity, considerable efforts have been made to find substitutes. Consequently, by 2020 there is expected to be a surplus of mercury in the world. Mercury exposure can cause serious health effects and a key strategy in reducing exposure is reduction in the use of mercury containing products and processes, efficient filtering when mercury or mercury compounds occur as by-products in industrial processes and disposal in a safe way to ensure isolation from man over long time periods.

Mercury and mercury containing waste will always remain toxic and hence are examples of wastes which require long-term safe storage. Because they maintain their toxicity over time, the isolation requirements needed for disposal of pure mercury and its compounds are of similar nature to those needed for disposal of spent nuclear fuel or long-lived radioactive waste from reprocessing.

A3.2.2 Health effects

Mercury has an impact on health on local, regional and global scales. Mercury and its compounds can be highly toxic to humans, ecosystems and wildlife. High doses can be fatal but also relative low doses can have serious adverse impacts to developing nervous system and there are indications of possible harmful effects on the cardiovascular, immune and reproductive systems.

The toxic risks from mercury depend on its chemical form, the manner of exposure, level and duration of exposure and vulnerability of persons exposed. The effects are increased by environmental bioaccumulation and biomagnifications through the food chain, especially through fish. In particular, mercury in the form of methyl mercury is hazardous to both humans and wildlife by ingestion as this compound passes the placental barrier and the blood-brain barrier. Elemental mercury is more toxic by the inhalation pathway.

Human exposure can result from several different pathways. Most important is the intake in food, primarily fish. Fish is an extremely valuable component of the human diet all over the world and mercury can be a major threat to this.

For elemental mercury, inhalation of mercury vapour that is then absorbed by lung tissue is the most important source in unhealthy working environments. To some extent, dental amalgam is another source of vapour. For other inorganic compounds, diet is the main source for exposure.

Many people are exposed to these ingestion and inhalation pathways. Their risks from mercury depend on a range of factors including employment, geographic location and diet, all of which contribute to determining levels of exposure.

Mercury has caused a variety of significant adverse impacts on human health and the environment throughout the world. The Minamata disease in Japan was caused by spilled mercury that converted to methyl mercury and bio-accumulated in fish and seafood that was the main source of food for local people. Around 3 000 people were affected. The case of Iraq mercury poisoning affected more than 6 000 people and was due to consumption of seed that had been treated with fungicides containing mercury.

A3.2.3   *Sources for releases and exposure*

The releases of mercury to the biosphere can originate from several different sources, as shown in Figure A3.2-1:

- natural sources – naturally mobilised from the earth's crust and also emissions from forest fires;
- impurities in raw material – anthropogenic releases related to mobilisation of impurities in fossil fuels, in particular coal, but also in oil and gas and also in the extraction of minerals;
- use of mercury in products and processes;
- re-mobilisation of historic mercury deposited in soil, sediments, water and tailings.

In order to cope with safety requirements over long periods, without the need for monitoring and intervention, the trend for managing long-lived hazardous waste is towards deep disposal. Several countries are developing such facilities.

Figure A3.2-1: Sources of mercury releases to the environment and the main control options
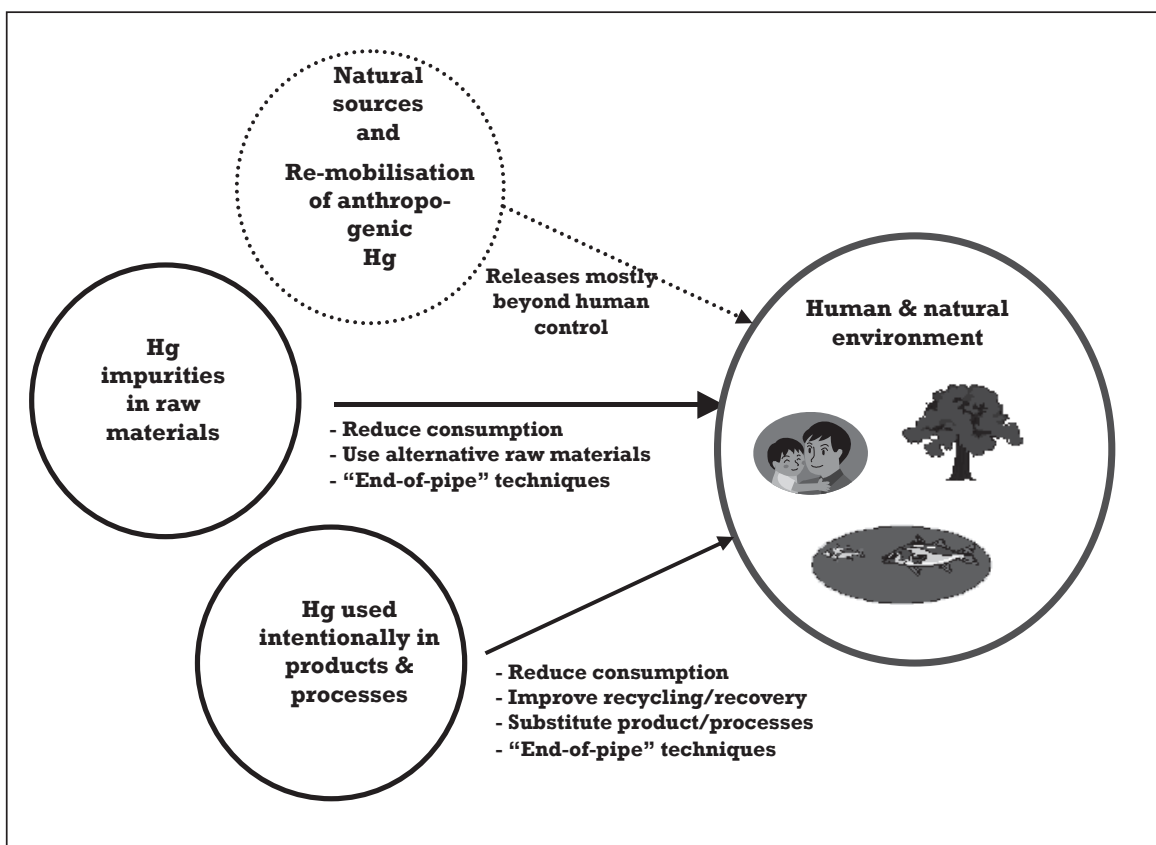
Source: UNEP, 2003.

A3.2.4 Amounts and cycling of mercury in the global environment

Mercury is available in soil and sediment in the ground, in water and air. In nature, mercury will change its properties and consequently participate in a number of biochemical cycles.

Possible routes for intake and damage are connected to its chemical form, methyl mercury being the most hazardous form. The most significant releases of mercury pollution are emissions to the air but mercury is also released from sources related to land and water.

Once released, mercury persists in the environment where it circulates between air, water, soil, sediments and biota in various forms. Thus, emissions add to a mobilised global pool of mercury that is deposited on land and water from where also will be re-mobilised. The time scale for the circulation between the different compartments contributing to the mobilised pool of mercury can be from some years up to thousands of years.

Estimates of the amounts of mercury include 5 000 t of mercury in the atmosphere, another 10 000 t in seas, 400 000 t in inland lakes and sediments and around 1 500 000 t in soil. The annual contribution to the mobilised pool has been estimated as 13 500 t.

A3.2.5 Efforts to reduce mercury releases and exposures

As local releases of mercury cause global problems, mercury is an issue much studied on global, regional, national and local levels. Despite reducing use and releases from industry, the emissions to air are increasing due to increased power production by fossil fuel combustion, especially coal. Artisan small-scale gold mining using mercury is causing huge health problems among native people in Asia, Africa and South America. To avoid damage to man and the environment, many improvements are needed.

Reduction of risks demands:
- reduced use of mercury in mining;
- efficient use of filters and other clean-up plant to avoid releases of impurities;
- collection, treatment and permanent disposal of mercury products and waste.

A3.2.6 Mercury waste – international activities

The United Nations Environment Programme (UNEP, 2003) carries out a comprehensive programme to understand mercury issues and to coordinate actions to reduce risks for humans and nature.

The *Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and Their Disposal* (Basel Convention) (UN, 1989) is the world's most comprehensive agreement on hazardous and other waste and aims to protect human health and the environment from inappropriate management of waste. A programme on mercury waste and its environmentally sound management is being carried out under the Basel Convention. Draft technical guidelines inform the practical steps needed to ensure sound and safe management.

The EU has a strategy and an active programme on mercury striving to reduce emissions and exposure, cutting supply and demand and looking for long-term disposal solutions including the support and promotion of international action such as within UNEP. Proposed legislation includes an export ban outside the EU and matters relevant to storage of surplus mercury.

The legal framework of EU concerns the following issues: (EC, 2003; EEC, 1991; 1999; EU, 2006)

- regulating releases into the environment (Directive 2006/11/EC) on releases to water environment;
- regulating wastes containing mercury (Directive 91/689/EEC) on hazardous waste;
- environmental standards for drinking water and foodstuffs such as fish;
- regulating storage and disposal (Directive 1999/31/EC) and (Decision 2003/33/EC) on disposal;
- a proposal (Regulation COD/2006/0206) on an export ban and for disposal of liquid mercury.

EU members are obliged to transpose and implement EU Directives into their own national legislations.

There is ongoing discussion in the EU aimed at revising (Directive 1999/31/EC) and (Decision 2003/33/EC) to allow future disposal of liquid elemental mercury in underground disposal facilities.

A3.2.7   Management of waste containing mercury

Mercury occurs in society in many forms from a large number of sources. Therefore, environmentally sound management of mercury is, in all respects, a complex task. In some industries, mercury is managed in a well-controlled manner whereas others are much less controlled. A variety of wastes such as gas filtering products, sludge from industrial processes, ashes and mineral residues, including used batteries and dental waste, is nowadays well looked after, at least from a short term perspective. Releases from historic waste, some coal power production and artisan gold mining are examples of areas that need to be improved.

Treatment of waste containing mercury

Hazardous waste, including mercury waste, is treated by a number of methods based on thermal, physical, chemical or biological processes. After collection and identification, the waste is sorted and packed in barrels, industry bags and containers for disposal.

Waste in powder form, materials from filters, sludge and similar products are often stabilised by being mixed with cement or fly ash. Recycling and reprocessing are used for batteries, contaminated soil etc, resulting in mercury in liquid form for storage and eventually disposal.

To dispose of surplus elemental mercury, methods have been developed to stabilise the liquid mercury by mixing with sulphur into a much more stable sulphide. Such products can be disposed of in hazardous waste landfills, on or in the ground, but not in an acid environment.

Disposal technology

Waste containing mercury is disposed of in general to specially engineered landfill, underground in caverns and pits close to the surface and deep underground in stable geological formations.

The bulk of waste containing mercury is disposed of in hazardous waste landfills, although historic waste may appear in many unqualified landfills. The disposal strategy and technology can differ significantly between countries.

For hazardous waste landfills in the EU, see Figure A2.4, requirements for design, safety and operation are stipulated in detailed directives implemented in the environment legislation of the

member countries. These landfills require monitoring and control of releases and are therefore not suited to long-term storage where such maintenance cannot be guaranteed.

Landfill in caverns and pits near the surface

Different types of chemical waste have been disposed of in caverns, excavated mine openings and pits and quarries near the surface. These allow better conditions for avoiding long-term leakage than surface landfills. In favourable geological situations, such facilities can be used for long-term safe disposal.

Underground landfills: disposal in deep geological formations

Disposal in deep stable geological formations is currently carried out in chambers situated in 700 m deep salt formations in Germany as shown in Figure A2.7. Several countries see such disposal techniques as the best and safest way available to manage long-lived hazardous waste (such as mercury containing waste). In Germany, large quantities of hazardous wastes – from Germany and some other European countries – are currently being disposed of in four mines.

The trend for the disposal of long-lived hazardous waste is toward such technology. Facilities are being developed in several countries to allow long-term safety without the need for monitoring and intervention.

Sweden was the first EU country, in 2005, to pass legislation requiring deep geological disposal for all waste with mercury content above 0.1%. To meet legislative requirements, Sweden is currently building a disposal facility in granite rock connected to a deep mine.

A3.2.8   Safety assessments

Although the basic principles are the same, the details of safety assessments for chemical and radioactive waste management are in general treated in different ways.

Although a few attempts have been made, there exists no common system to evaluate risks. From the viewpoint of society, it is desirable to judge risks in a way that can be applied to both categories. Some attempts to discuss an "overall risk" have found it useful to separate effects leading to cancer from those that have other serious effects on health. Hazardous waste exhibits a range of characteristics that have serious effects on health, including explosive, flammable, oxidising, poisonous, infectious and toxic. These tend to be "non-cancer" risks. The primary hazard from radioactive waste is exposure to radiation, which can lead to cancer.

However, the boundary is not always clear, as some toxic chemicals can cause cancer and some compounds that are radioactive are also toxic. A primary risk from uranium in drinking water, as an example, is from its toxicity to the kidney.

Management and disposal of waste containing mercury and its compounds is regulated through national regulations for hazardous materials that derive, in general, from EU Directives and Basel Convention statements.

Safety regulation is however focussed on temporary storage and monitored disposal over short time periods – 30 to 200 years. The long-term safety assessments required for final safe disposal of mercury and mercury waste are in general only briefly mentioned in the regulations of most countries.

The EU Directives give requirements and guidance on issues related to geological repositories and requirements on safety assessments for licensing and use. These requirements ask for consideration of waste characteristics, the technology used and in particular the geological properties of the repository. In most aspects, requirements to demonstrate safety are of a similar nature to those stipulated for long-lived radioactive waste. However, the Directives are less detailed on the time periods to be considered, mentioning thousands of years or geological time periods. Safety assessments for the licensed disposal facilities in deep salt mines in Germany deal with the long term by stating that the geological conditions of the salt formation itself provides stability and containment over millions of years.

A3.2.9 Attitudes of the public, politicians and regulators

The public's attitudes and perception of risks are different for hazardous waste and its disposal if the waste has a toxic chemical content or if it is radioactive, see Appendix 4.

However, regulators are active in both areas and requirements on polluting industry and disposal are stringent for both categories of wastes.

A3.2.10 Comparison with radioactive waste

Occurrence, exposure and health effects

Mercury and its compounds are highly toxic and present risks to human health and the environment over long periods that require precautions that are similar in some ways to those needed for long-lived radioactive waste, particularly safe permanent disposal. In both cases, releases are often local but the impacts can be on a global scale if releases are to the atmosphere.

The annual global contribution to the mobilised pool of mercury has been estimated as 13 500 tonnes. To provide a perspective, this amount is in the same order of magnitude as the annual global spent fuel arising from nuclear power plants, which is estimated to be about 15 000 tonnes. However, the hazards from the two waste types are, of course, very different. Mercury mobilised by man is distributed around the globe in relatively small concentrations, but with the potential to affect the health of very large numbers of people. Spent fuel is securely contained in a limited number of locations with the potential to affect only a small number of people, and then only in the event of a very low probability accident.

Safety

Safe management and disposal must be demonstrated in both the short and the long term for waste containing mercury and for radioactive waste. Because mercury is stable it will always be a risk to human health and the environment, and the very long-term scenarios are even more important than for radioactive waste, where decay will eventually reduce the risk (albeit the timescale for the activity in spent fuel to decay to around the level of the original uranium ore is around 100 000 years).

In both cases, regulations regarding tolerable releases (radiation dose, content of mercury in fish/water, etc.) and short-term issues are well established. Compared with the large R&D programmes for the long-term management of radioactive waste, corresponding management of mercury waste is currently less well studied.

Final disposal

Currently, final disposal of mercury waste is carried out in landfills, particularly engineered facilities for hazardous waste, and in stable geological formations, primarily deep salt mines. As the landfills must be monitored and managed the trend is toward disposal in stable geological formation where there is less need for institutional control in the long term. The best examples are salt mines.

State of knowledge

Comprehensive R&D is carried out for management of both radioactive and hazardous waste. However, the level of data collected and resources spent are higher for radioactive waste. Considering the number of chemical substances to be addressed, R&D resources must be directed to a much broader range of problems in the case of hazardous waste and are not primarily directed towards final disposal.

Legislative and regulatory framework

Comprehensive and detailed regulation and legislation exists for the management and disposal of both mercury and radioactive waste.

Regulation concerning mercury waste, by being a part of overall environment legislation, is more general and harmonised on both regional and international levels. On the international level, UNEP and the Basel Convention explore the needs and give recommendations for efficient and environmentally sound management. EU regulation and legislation stipulates requirements for management and disposal within EU. The EU regulation is in turn mandatory for member states and must be implemented in national legislation.

Regulation and legislation on management and disposal of radioactive waste is also based on very active international cooperation but matters are finally decided and regulated in specific national legislation.

A3.3 Potential future management of CO_2: carbon capture and storage (CCS)

A3.3.1 Background

Worldwide concern over human-induced climate change has led to the signing of the Kyoto protocol whereby Governments have made binding commitments to reducing greenhouse gas emissions. In addition, the introduction of carbon trading provides an economic stimulus to reduce fossil fuel usage. Governments are pursuing a number of parallel policies in their attempts to fulfil their Kyoto obligations. These include energy conservation and subsidies to producers and users of renewable energy devices. Governments are also investing in research into ways of reducing the carbon footprint of the more traditional means of electricity generation, especially the burning of coal and other fossil fuels. Foremost amongst the proposed solutions is carbon capture and storage (CCS, Figure 1). This technology will necessarily impose penalties in terms of additional cost and additional energy usage. As with new-build nuclear power, critics argue that it is a distraction from the need to invest in the development of renewable energy sources.

In line with current practice in the carbon capture and storage business, the word "storage" is used throughout this section of Appendix 3. It is interesting to note the contrast with the terminology used in radioactive waste management where "storage" always implies an intention to retrieve and where, if there is no intention to retrieve, the word "disposal" is used. Similarly, in carbon capture and

storage, CO_2 is never referred to as "waste" – another difference from radioactive waste management perhaps recognising that, when used for enhanced oil recovery, it is a useful product. Enhanced oil recovery, a process whereby CO_2 is injected into diminishing oil reservoirs to boost production, has been in routine use for more than 30 years.

A3.3.2   *Sources and amounts of current release*

The International Panel on Climate Change (IPCC, 2007) states that emissions of the greenhouse gases covered by the Kyoto Protocol were 49.0 Gt of CO_2-equivalent (eq.) in 2004, an increase of 24% since 1990. The largest fraction (29 Gt) was from carbon dioxide (CO_2) itself. Electricity generation is by far the largest and fastest growing source of CO_2. Around 40% of global primary energy was used as fuel to generate 17 408 TWh of electricity in 2004 with about 67% of this being fossil fuelled.

IPCC (2007) estimates that, when applied to both coal- and gas-fired electricity generation, CCS could result in a 0.81 Gt CO_2 eq. total reduction in greenhouse gas emissions by 2030. This is broadly similar to the figures for hydro and wind (0.87, 0.93 Gt CO_2 eq. respectively). Emission reductions from applying CCS to coal-fired generation are estimated to be 0.49 Gt CO_2 eq. IPCC estimates that nuclear energy could reduce emissions by a further 1.9 Gt CO_2 eq. beyond the 1.7 Gt CO_2 eq. already anticipated by reference to IAE's World Energy Outlook 2004. (IEA, 2004a)

A3.3.3   *Carbon capture*

Carbon capture (IEAGHG, 2007) requires a very significant investment so that the technology is only suitable for large producers of CO_2. Primarily, these are fossil-fuelled electricity producers (emitting 10.5 Gt CO_2 per annum) and, to a lesser extent, cement manufacture, refineries, steel production, etc. (IPCC, 2005) A single 1 600 MW lignite-fuelled power station emits around 10 million tonnes of CO_2 per year. (Vattenfall, 2008)

CO_2 capture technology can be deployed to good effect with combined cycle gas turbine plant. CCGT have high thermal efficiency and may burn either natural gas or hydrogen and carbon monoxide produced from coal. The fact that the fuel is, or is made to become, gaseous allows the possibility that CO_2 may be captured either before or after combustion.

The pre-combustion method is used with coal-fired CCGT where, in the absence of CO_2 capture, proprietary compounds such as Selexol are is used to remove sulphur oxides from the H_2 and CO gas mix prior to combustion. These compounds will also remove CO_2 although, in a normal coal-fired CCGT, this is an unwanted reaction. If CO_2 capture is wanted, however, oxidation of the coal during gasification is allowed to go a little further to produce hydrogen and CO_2 so that the latter may be removed.

Most conventional coal power plants burn pulverised coal and would, therefore, need post-combustion capture technologies. The UK government, for example, is specifically supporting this option because of its application to China and other emerging economies with large numbers of conventional coal power plant. There are two post-combustion methods. In the first, the CO_2 is removed from the flue gas by means of a chemical or physical reaction. Most often, proprietary organic compounds (based on amines) react chemically with the CO_2 and are then regenerated by reaction with steam. CO_2 can then be cooled, dried and pumped away. A complication with this method is that steps must be taken to remove the oxides of sulphur and nitrogen so that they cannot react with the organic chemicals. If they do, they will form stable products that prevent the organic compounds from being regenerated.

Figure A3.3-1: Outline scheme illustrating carbon-free electricity generation from fossil fuels using terrestrial or marine-based geological storage
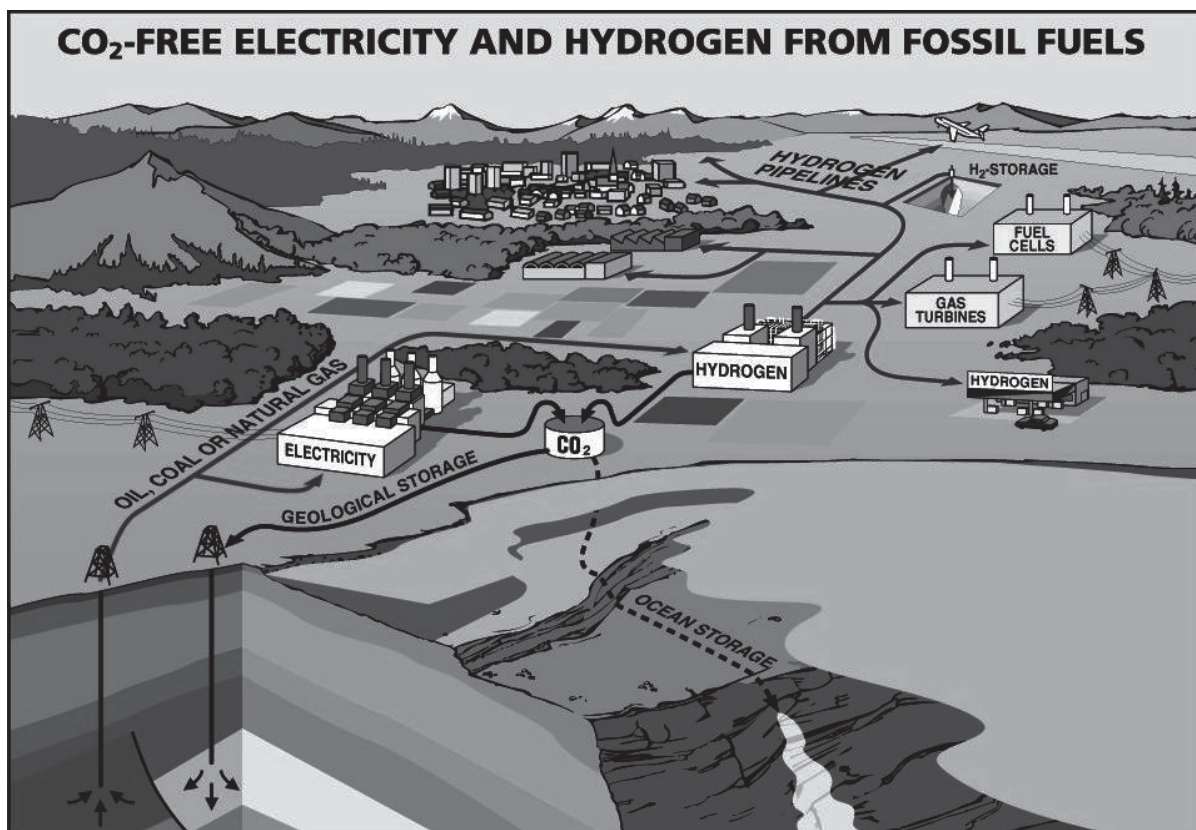
Source: IEAGHG, 2007.

The second form of post-combustion CO_2 capture is known as oxy-combustion. This, again, may be used with conventional pulverised coal plant if the coal is burned in pure oxygen. The oxygen is produced on-site using an air separation plant. The flue gas consists almost entirely of water and CO_2 so that post-production processes can be conducted with higher efficiency. A possible offset against the cost of air separation is the fact that the flue gas may need little cleanup. This is because sulphur oxides are removed with the CO_2 and burning in oxygen results in the flue gas having low levels nitrogen oxides. Note, however, that the pilot CCS plant at Spremberg in Germany does have flue gas desulphurisation.

A3.3.4 Principles of CO_2 storage

All current underground storage designs aim to store the CO_2 at a depth of greater than 800 m because these depths produce a pressure at which CO_2 exists in a supercritical state. (IEAGHG, 2008a) A supercritical state is one in which the material is neither liquid nor gas but, rather, behaves like both. The advantages are twofold: there is a volume reduction (compared to the gas at room temperature and pressure) of at least 200 times and the supercritical CO_2 can flow easily (like a gas) into the pore spaces between mineral grains in the host rock.

Using natural gas fields as an analogue, the general argument is that rock formations are capable of containing gases for millions of years. Mechanistic explanations are available that explain how the gas comes to be trapped and why there is reason to believe that trapping will be permanent (see Box 1).

BOX 1: Trapping mechanisms

Trapping of CO_2 occurs by four different mechanisms (IEAGHG, 2007):

 — stratigraphic/structural;

 — residual;

 — solubility;

 — mineral.

Stratigraphic and structural trapping refer to large-scale geological features that allow gas or liquids to be trapped underground. Almost invariably, this arises because an impermeable formation lies above a reservoir formation as a result of the stratigraphy or as a result of some disturbance to the stratigraphy due to faulting (see figure right).
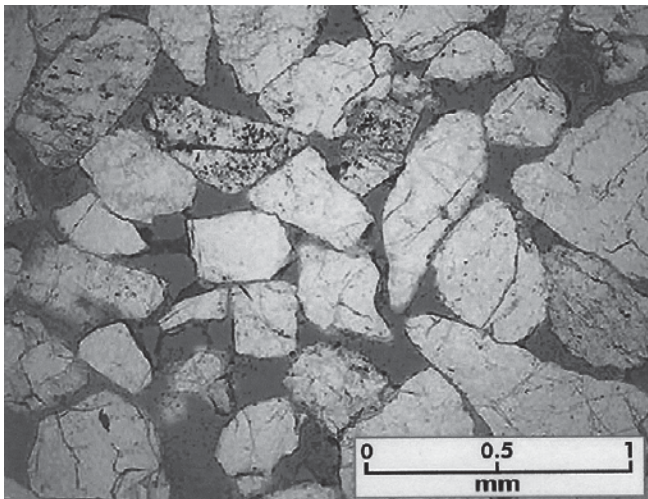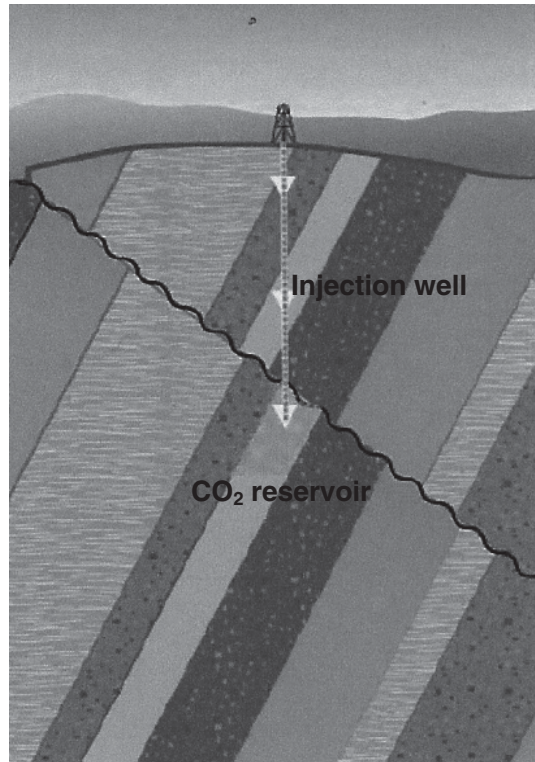
During CO_2 injection, the applied pressure must be high enough to allow the CO_2 (which appears blue in the figure left) to displace formation fluids (e.g. water or oil) from the rock pores. At the same time, the pressure must be not so high as to break the stratigraphic or structural seal. When injection stops, the pressure drops and the surrounding fluid moves back into the pores (propelled by capillary action), trapping the CO_2 – this is known as **residual** trapping.

CO_2 may then dissolve in the water (**solubility** trapping) forming a more dense fluid that may slowly sink through the formation. Over thousands of years, the dissolved CO_2 may react with the surrounding minerals to form solid products (**mineral** trapping). The timing of these processes means that CO_2 trapping becomes more secure with time, and hence the risk of leakage decreases with time. (IPCC, 2005)

Source of images: CO2CRC.

A3.3.5   Cost of CCS

CCS places additional energy demands, principally from separation and compression. Depending on the type of plant and the nature of the fuel, a power plant equipped with CCS would need roughly 10-40% more energy than an equivalent plant operating without CCS. The additional energy requirement will itself produce CO_2 and the net result is that a power plant with CCS should reduce CO_2 emissions to the atmosphere by approximately 80-90% compared to a plant without CCS. (IPCC, 2005)

Figures presented by IPCC (IPCC, 2005) indicate that carbon capture alone increases the cost of electricity by:

- 1.8 to 3.4 US$ct per kWh for a pulverised coal power plant;
- 0.9 to 2.2 US$ct per kWh for an integrated gasification combined cycle coal power plant;
- 1.2 to 2.4 US$ct per kWh for a natural gas combined-cycle power plant.

Transportation and storage would add between -1 and +1 US$ct kWh^{-1} and about half this for gas plants. The negative figure recognises the revenue that would arise if CO_2 were used for enhanced oil recovery. If we (i) ignore the highest capture costs (for a pulverised coal plant); (ii) assume that transport and storage are cost-neutral; and (iii) take a mean wholesale cost of electricity of 4 US$ct per kWh, these figures represent a percentage increase in the cost of electricity of between 22 and 60%.

A3.3.6   Suitable geological formations

According to the IPCC, the potential storage capacity in geological formations worldwide far outstrips the likely demand. The main requirements of a CO_2 storage site (using the standard industry terminology) are: (IEAGHG, 2008a)

- accessibility – a geological formation that is accessible by borehole;
- capacity – the ability to hold useful quantities of gas;
- injectivity – the speed with which the formation can receive gas;
- storage security – leak tightness of the formation.

Many geological formations are thought to meet these needs but the current front runners are:

- depleted oil and gas reservoirs;
- deep saline formations;
- un-mineable coal seams.

Depleted oil and gas fields will probably be the first sites to be used for CO_2 storage because of their known location, their known properties, their availability and the greater certainty with respect to the underlying science. CO_2 injection is already used as a means of enhancing oil and gas recovery and it is possible that such enhanced recovery could be a means of offsetting the cost of storage. On the other hand oil and gas fields will not usually be located close to the CO_2 production sites and there may be concerns that abandoned wells may not have been sufficiently well sealed to ensure leak-tightness.

In the longer term, the extremely wide distribution of deep saline formations will probably allow them to constitute the majority of CO_2 disposal sites. A possible limiting factor is that these formations may not always occur at a convenient depth: either too deep, which will increase cost, or too shallow, which will not allow CO_2 to reach the supercritical state. This type of geology should have good long-term retention properties for CO_2 although stratigraphic/structural trapping (Box 1) may not

always be as obviously present as it is for former oil or gas reservoirs. Abandoned wells are less of an issue than for former oil and gas reservoirs but, still, cannot be wholly dismissed.

Un-mineable coal seams are a more distant prospect: it is known that coal can hold significant quantities of gas in micropores but the mechanisms are imperfectly understood at present. An advantage of these formations is that the cost of injection could be offset if the CO_2 displaced methane, which could then be extracted for use as fuel.

A3.3.7   Pilot projects

As already noted, CO_2 is routinely injected into oil reservoirs for the purpose of enhanced oil recovery. Typically, natural gas (methane) is pumped to an installation where it is partially oxidised or "reformed" to create hydrogen and CO_2. The CO_2 is then separated and pumped to an oil well whose production is diminishing. The CO_2 boosts oil production by displacing oil from the reservoir formation. These arrangements appear to form the basis of many of the 50 or so completed, ongoing or planned pilot projects for CO_2 storage worldwide. (SCCS, 2008) Three projects are particularly noteworthy for their size. The Weyburn-Midale CO_2 storage and monitoring project in Canada injected more than 5 Mt of CO_2 into a depleted oilfield. The CO_2 is supplied from a coal gasification plant in North Dakota, United States. An extensive monitoring network failed to detect any leakage. In the Sleipner project, 10 Mt of CO_2 have been injected into a deep saline formation off the Norwegian coast. (IEAGHG, 2008a) The Krechbah processing plant in Algeria has, since 2004, re-injected 1.2 Mt CO_2 per year into the gas field it came from.

There appears to be only one operational project that is attempting to demonstrate both carbon capture and storage. This is a 30 MW(e) coal-fired oxy-combustion plant near Spremberg in Germany. CO_2 is collected, compressed and trucked 350 km to an empty gas field for injection. It is expected that 100 000 t of CO_2 will be injected over 3 years. The plant has been funded by Vattenfall (the Swedish power generator) at a cost of 70 M . Interestingly, the flue gas is cleaned to remove sulphur dioxide and fly ash. Other projects are being proposed and their feasibilities investigated around the world. In particular, the EU ZEP programme (Zero Emission Fossil Fuel Power Plants) (EU, 2008) aims to have up to 12 large scale CCS projects operational by 2015 so as to demonstrate commercial viability by 2020.

A3.3.8   Risk assessments

Risk assessments are used in the oil industry to demonstrate the safety of CO_2 injection for enhanced oil recovery. Increasingly, methodologies developed for radioactive waste disposal are being used to assess long-term effects. For instance, assessments commonly use base (normal) and alternative scenarios to address possible future states of the storage and the surrounding environment. Similarly, standardised lists of features, events and processes (FEPs) may be used for auditing assessments and there is frequent reference to natural analogues and site-specific analogues such as groundwater residence times. Box 2 describes the approach to, and the lessons drawn from, risk assessment in the Weybourn project. (IEA, 2004b)

The unresolved issues identified in the Weybourn project are characteristic of safety assessment in radioactive waste disposal: typically, they hinge on the need for the assessment model properly to represent the disposal environment and, in particular, for the model to explain the characteristic features of the host rocks.

Other similarities include the need to assess seismicity and vulcanism, geochemical effects (including the action of CO_2 on repository seals) and the effects of minor constituents on repository behaviour.

IEAGHG (2008a) points to the many monitoring techniques available to verify the amount of CO_2 injected and the integrity of the storage. As with radioactive waste disposal, monitoring is of limited use when attempting to verify long-term containment but this is less of an issue in Carbon dioxide Capture and Storage (CCS) because leakage is most likely during or soon after injection so that CO_2 storage becomes more secure with time. Consequently, IPCC guidance, London Dumping Convention, OSPAR Treaty and EU CCS Directive all allow monitoring to decrease with time and cease if all evidence indicates secure storage.

A3.3.9 Regulation

CCS is a new technology and regulation is evolving. The IPCC special report on CCS states that: (IPCC, 2005)

> "Existing laws and regulations regarding inter alia mining, oil and gas operations, pollution control, waste disposal, drinking water, treatment of high-pressure gases and subsurface property rights may be relevant to geological CO_2 storage. Long-term liability issues associated with the leakage of CO_2 to the atmosphere and local environmental impacts are generally unresolved."

According to Vattenfall (2008), responsibility for post-injection monitoring (and, presumably, remediation, if monitoring found something untoward) could rest with the operator, the government, a third party brought in for the purpose or any combination of these. As with radioactive waste disposal or abandoned mines, governments will invariably be the long-term guarantors of safety. The key issue for operators (for which read investors in CCS) will always be the duration of the operator's responsibility.

An essential precondition for development of CCS is the ability to profit from reduced CO_2 emissions. The IPCC Greenhouse Gas Inventory Guidelines (2006) provide a methodology for assessing the effect of CCS on greenhouse gas emissions, thus enabling countries to report emissions reductions in their inventories from CCS, and providing the basis for its inclusion in emissions trading schemes. The EU Greenhouse Gas Emission Trading Scheme (ETS) started allowing trading in CCS emission reductions in 2008.

With respect to sub-sea storage of CO_2, the London Dumping Convention and its 1996 Protocol applies; the parties to the Protocol agreed in 2006 to permit sub-seabed storage of CO_2. OSPAR did the same in 2007.

The risk assessment performed for the Weyburn project addressed five possible release scenarios (IEAGHG, 2007).

1. *Rapid "short-circuit" release* (via fracture, borehole, or unconformity). Typically, short circuit releases would cause acute environmental or health effects such as might be produced by high concentrations of CO_2 in low-lying areas on the surface. The presence of unknown or poorly sealed wells penetrating into the storage formation is generally considered to be the most important release pathway.

2. *Potential long-term release*. Long-term releases may be impossible to measure but are important because they determine the overall effectiveness of CCS.

3. *Induced seismic event*. Induced seismicity was first seen in the 1960s at some underground storage sites for natural gas. Raised gas pressure allows small movements (micro-seismicity) along active faults. Since then storage sites have aimed to avoid active faults but even so, it is necessary to have an understanding of the process and to know, for example, how high the gas pressure needs to be to trigger such an event.

4. *Disruption of host rock*. As with the induced seismic event, it is important to understand how gas pressures might cause failure of the sealing formation and to know how large the gas pressure needs to be to cause such an effect.

5. *Release to aquifer*. This is an important issue not least because regulations are often framed in terms of maintaining groundwater quality. Risks to shallow water aquifers may arise from acidification, unwanted mineralogical effects and upwards displacement of briny waters.

As a result of the assessments, issues requiring further development were identified. These include:
- the use of more direct monitoring to demonstrate effective storage;
- more effective use of existing seismic data;
- determine the fate of gaseous impurities: H_2S and mercaptans;
- characterise conductive natural fractures in strata overlying the reservoir (if they exist) and their flow properties;
- obtain core samples to determine mechanical properties of any weakened overlying/underlying strata and properly preserve;
- assess the impact of fractures on seismic images (anomalies may be due to more than the presence of CO_2);
- in long-term fate assessment, account for additional mechanisms that may dissolve reservoir rock or minerally fixate CO_2 (e.g. dissolution due to convective mixing) and perform sensitivity analyses for various long-term assessment models.

A3.3.10 Attitudes of public, governments and regulators

The IEA acknowledges (IEAGHG, 2008b) that public acceptance will be needed if CCS is to progress and IPCC frequently mentions its importance. (IPCC, 2005) Few public opinion surveys have been conducted (Tokushige, *et al*, 2007) and these few have not been given wide publicity. Neither of the two largest CO_2 storage projects (Weyburn and Sleipner) have public acceptability as part of their remit. Given that CO_2 injection is already used as a standard method of enhanced oil recovery, it is possible that the CCS industry considers that public acceptability is unlikely to be a "show stopper". Anecdotal evidence from Spremberg (a coal mining town), where the pilot CCS plant is located suggests that the public broadly approves of the project with comments like "It's bound to bring jobs, that's what matters, but if it makes us famous for saving the world, that would be cool". (Smith, 2008)

Green groups vary in their view of CCS. Friends of the Earth International (FoE) classes CCS and nuclear energy alike: as "unsustainable technologies" (FOEI, 2005), though some national FoE groups may be more accommodating in their approach. Greenpeace International opposes the

application of CCS to coal-fired power stations as a means to combat climate change. (Greenpeace International, 2007) WWF is in favour of CCS, but does not support the Clean Development Mechanism (CDM), an arrangement under the Kyoto Protocol that allows certain countries to invest in projects that reduce emissions in developing countries as an alternative to more expensive emission reductions in their own countries.

Governments face a dilemma: increasing domestic demand for electricity coupled with a need (or even binding commitments) to reducing CO_2 emissions. It is clear that no single measure, whether energy saving, renewable electricity sources or nuclear power will solve the problem. In this situation, governments will aim to adopt a wide range of measures in parallel; these measures will include CCS. President Bush, for instance stated in 2001: "We all believe technology offers great promise to significantly reduce [greenhouse gas] emissions – especially carbon capture, storage and sequestration technologies."

As one might expect, regulators appear to be content to regulate CSS provided that they have the necessary powers and funding. It is clear that many regulators are informing themselves about CCS and (presumably) assessing the need for new regulations. US EPA say that it aims to ensure that geological sequestration does not endanger underground sources of drinking water. The US regulations cover well siting, well construction, well operation, and well closure and there have been over 800 000 regulated wells injecting a variety of fluids over the past 30 years. The EC DG Environment has proposed a Directive to create an enabling legal framework in the EU and to remove existing regulatory barriers.

In responding to the UK Government announcement of new coal-fired power stations, the Environment Agency (responsible for waste disposals in England and Wales) goes further stating that: "new and replacement coal-fired power stations should only be permitted where they are capable of capture and storage of carbon dioxide"; and "the Environment Agency can help to assess all new plant, subject to an appropriate role and funding".

A3.3.11 Discussion and conclusions

General differences and similarities with radioactive waste disposal

The main differentiating feature between radioactive waste disposal and CCS lies in the nature of the disposed material. In the case of CCS, the stored CO_2 is simple chemically but complicated physically since it may exist as a liquid, a gas or neither (i.e. it may be a supercritical fluid) and these different phases may be simultaneously present in different parts of the storage system. It also has very high volume. The phase changes make the system difficult to model, and the large volumes have the potential to affect the evolution of the system. For radioactive waste disposal on the other hand, the waste inventory may be complicated chemically but it is predominantly composed of solid material. Furthermore, the overall waste volumes are relatively small and radionuclides are present only in trace amounts so that, with the possible exception of alkaline plumes emanating from cement-based repositories, radioactive waste disposal does not greatly affect the natural evolution of the system.

Another point of difference is that, in general, emplacement of solid radioactive wastes is intended to be performed in underground facilities whereas CCS is intended to be performed from the surface using boreholes. Both technologies have advantages and disadvantages: disposal from the surface will clearly be cheaper but it will also hinder detailed characterisation of the repository host rocks both in their natural state and in post-injection.

In searching for a suitable site, there are, once again, similarities and differences. Both technologies would try to avoid seismically and volcanically active areas. Both would also aim to understand the evolution of the site so that the past might be used as a guide to the future. However, whereas radioactive waste disposal usually aims to combine engineered and natural barriers to contain the radionuclides in the waste, CCS uses only natural barriers. So, for instance, a repository for spent nuclear fuel may place the spent fuel inside steel or copper canisters while a repository for intermediate-level wastes may use large quantities of concrete. With the exception of the seal to the injection well, a geologic storage for CO_2, would not use such methods.

Another possible difference is that radioactive waste disposal would generally try to avoid so-called "complex sites". This may not be an option for CCS given the large number of sites needed and, indeed, the geology of some pilot project sites may be regarded as complex (e.g. Weybourn).

Safety assessments

In developing appropriate risk assessments, CCS appears to have borrowed widely from safety assessment methodologies for radioactive waste disposal. Consequently, we find familiar approaches such as the use of scenarios to encompass possible future states of the repository and its surroundings; standardised lists of features, events and processes (FEPs); and natural analogues.

In assessing long-term impacts, radioactive waste disposal generally has very well defined calculational end points that are directly derived from numerical limits and constraints imposed by regulators. An example is the annual radiation dose to an exposed individual that can be traced back to documents such as the Basic Safety Standards. (IAEA, 1996) It seems that there is no such universally adopted measure of health detriment for CCS risk assessments but, rather, a wide range of human and environmental safety issues that are not always precisely defined.

Indicative costs

Accurate cost estimation is difficult and the simplest method, perhaps, is to compare the additional costs of disposal in terms of the premium that needs to be placed on the cost of electricity generation.

In the case of radioactive waste disposal, the cost probably ranges between 5 and 10% of the cost of electricity. As described above, the add-on costs of CCS range between 22 and 60% mostly depending on the type of plant.

State of knowledge

The US DOE (2008) calls for further work to show that CCS:

- is effective and cost-competitive;
- provides stable, long term storage; and
- is environmentally benign.

Examining these in turn, US DOE states that using present technology, sequestration costs are in the range of 100 to 300 USD/ton of carbon emissions avoided. The goal of DOE's programme is to reduce this to 10 USD or less by 2015.

Storage of natural gas in underground formations has been practised for around 100 years while CO_2 injection for the purpose of enhanced oil recovery has been performed for almost 40 years. From these it is clear that CO_2 can be stored in deep underground formations without detectable losses over these timescales. It seems, however, that the accuracy of the measurements is not sufficiently high to

provide confidence for CO_2 retention in the long term – evidence for this is more general, coming from natural analogues. In developing a methodology to allow specific CCS schemes to claim credit under the Kyoto Protocol, the IPCC has made allowance for this uncertainty. (IPCC, 2006)

The final issue, environmental safety, is discussed above.

Legislative framework

As noted above, some countries already have regulations controlling CO_2 injection for enhanced oil recovery. No doubt, these will form the basis of regulations that address long-term retention of CO_2 also. In the long term, only governments can bear the liabilities that might accrue from failure of CO_2 storage. The crucial issue for operators and investors in CCS is the timing of the changeover from a private to a public liability.

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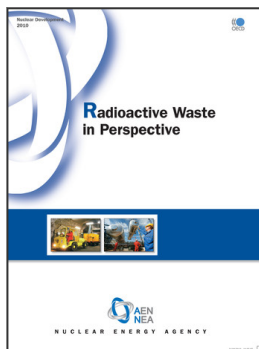
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