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HEALTH AND ENVIRONMENTAL EFFECTS OF COAL-FIRED ELECTRIC POWER PLANTS

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ABSTRACT

This paper describes health and environmental impacts of coal-fired electric power plants. Effects on man, agriculture, and natural ecosystems are considered. These effects may result from direct impacts or exposures via air, water, and food chains. The paper is organized by geographical extent of effect. Occupational health impacts and local environmental effects such as noise and solid waste leachate are treated first. Then, regional effects of air pollution, including acid rain, are analyzed. Finally, potential global impacts are examined.

Occupational health concerns considered include exposure to noise, dust, asbestos, mercury, and combustion products, and resulting injury and disease. Local effects considered include noise; air and water emissions of coal storage piles, solid waste operations, and cooling systems. Air pollution, once an acute local problem, is now a regional concern. Acute and chronic direct health effects are considered. Special attention is given to potential effects of radionuclides in coal and of acid rain. Finally, potential global impacts associated with carbon dioxide emissions are considered.

HEALTH AND ENVIRONMENTAL EFFECTS OF COAL-FIRED ELECTRIC POWER PLANTS*

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1. INTRODUCTION

This paper describes health and environmental impacts of coal-fired electric power plants. Effects on man, agriculture, and natural ecosystems are considered. These effects may result from direct impacts or exposures via air, water, and food chains. The paper is organized by geographical extent of effect. Occupational health impacts and local environmental effects such as noise and solid waste leachate are treated first. Then, regional effects of air pollution, including acid rain, are analyzed. Finally, potential global impacts are examined.

Typical, modern, coal-fired electric power plants are of ~1 GWe capacity with availability of 73% and net efficiency of 35% (EPRI, 1982). For simplicity, quantitative figures will be presented on a GWe-y basis. Plants of this size consume about 10,000 metric tons of coal daily, requiring massive material handling equipment.

There are a wide variety of health and environmental effects to be considered, environmental routes through which they occur, and additional considerations such as whether they are temporary or permanent and whether they occur quickly or only after an extended time. Matrices have been developed to provide a general checklist or reminder of the full range of

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impacts that should be considered (Leopold et al., 1971; Brubaker, 1972). In addition, systems approaches to environmental impact which trace chains of environmental perturbations from emissions through physical-chemical systems and plant-animal systems to man provide the basis of an approach to assessment (Morris and Novak, 1976).

2. OCCUPATIONAL HEALTH EFFECTS

Large U.S. coal-fired power plants (over 100 MWe) have an average labor force of 490 persons (EIA, 1980). These workers include asbestos workers, conveyor-dumper car operators (coal handlers), boilermakers, steamfitters, machinists, electricians, operators and assistant operators, instrument mechanics, and laborers (TVA, 1981). Workers are potentially exposed to noise, heat stress, coal dust, solvents, asbestos, mercury, and combustion products (fly ash, sulfur dioxide, oxides of nitrogen, carbon monoxide). Switch yard workers have potential exposure to electromagnetic fields. Data on these exposures are limited and will be drawn primarily from a study of two Tennessee Valley Authority plants (TVA, 1981). A national survey conducted by the National Institute of Occupational Safety and Health, which included 2,673 power station operators found 80% were exposed to continuous noise and more than 23% to trichlorethylene and other solvents (Rom, 1980).

Noise. In the TVA study, geometric mean 8-hour noise exposure levels (Table 1) were as high as 126 dBA for auxiliary operators. In these plants, hearing protection must be worn in high noise areas, so these exposures are not necessarily the doses actually received by workers.

Table 1
Geometric Mean 8-Hour Noise Exposures in Coal-Fired Power Plants

Assistant Unit Operators	80 dBA
Auxiliary Operators	126
Heavy Equipment Operators	92
Laborers	77

Source: TVA, 1981.

Coal Dust. Coal conveyor and dumper car operators had a geometric mean exposure to respirable coal dust of 0.9 mg/m^3 , as measured by over 700 8-hour samples taken over 6 years at TVA plants. Fewer samples were available for heavy equipment operators working on coal storage piles, but respirable coal dust measurements for these workers ranged from <0.1 to 6.8 mg/m^3 with a geometric mean of 0.5 (TVA, 1981).

Asbestos. Much of the insulating material in older plants was asbestos. Since the early 1970s there have been intensive efforts to decrease asbestos exposures. TVA personal sampling data from 1973-79 show a decrease from a geometric mean of 1.1 fibers/ml to 0.5 over the period. The samples are not average exposures, but were taken only when known asbestos work was being conducted. There is, however, no trend in the number of samples (TVA, 1981).

Mercury. Instrument mechanics and employees cleaning up mercury spills are potentially exposed to mercury vapor. Airborne concentrations of mercury in TVA coal-fired power plant instrument shops ranged from not detectable to 0.5 mg/m^3 , although the total number of samples was small. Urinary mercury levels in employees cleaning up mercury spills ranged from not detectable to 41 ug/l (TVA, 1981).

Combustion Products. In the TVA study (TVA, 1981), exposures to respirable fly ash were 0.2-0.5 mg/m³ with all craft groups within this range. Personal monitoring of sulfur dioxide and nitrogen dioxide showed no appreciable levels, but statistical extrapolation suggested that electricians and operators could be exposed to levels of sulfur dioxide exceeding 5 ppm (the occupational exposure limit) in 7-8% of samples. Area measurements as high as 40 ppm of nitrogen oxide were found in one plant, but follow-up personal monitoring found over half the samples <1 ppm and the highest to be 5 ppm. Concentrations of carbon monoxide were found to be up to 14 ppm in combustion gas leaks, but normal work areas were 0-2 ppm.

Occupational Injury. The Council on Environmental Quality estimated occupational injuries as 0.1 deaths and 1.2 injuries per 1000 MWe plant year for coal-fired electric power plants (CEQ, 1970). These are equivalent to 0.13 deaths and 1.6 injuries per GWe-y. Based on survey results reported by Bertolett and Fox (1974), Morris (1983) estimated non-fatal injury rates in coal-fired power plants to be 4.4 per GWe-y. More recent Canadian data show 7.8 non-fatal injuries per GWe-y in fossil plants (Hamilton and Wilson, 1982). Numbers of deaths are small and definitions of non-fatal injuries vary internationally and over time.

Occupational Disease. Ontario hydro has sponsored an epidemiological follow-up of their workers and pensioners. The usual healthy worker effect appears. The standard mortality ratio for cancer was slightly elevated, but was not statistically significant. The possibility was suggested that this may represent exposure to asbestos in early years (Hamilton and Wilson, 1982).

3. LOCAL EFFECTS

Potential local effects include noise; air pollution from coal pile blow-off, cooling towers, and stacks; ground water contamination; and thermal and chemical pollution of streams or lakes. Land use, socio-economic, and aesthetic impacts are not discussed. Most air pollution impacts are included under regional effects.

Noise. Sources of noise from coal-fired power plants include the turbines, generators, and associated equipment; cooling towers; coal handling machinery; and large trucks and trains. Noise measurements about 300 m from the plant edge of three coal-fired plants under full load ranged from 49.3 to 60.7 dBA (Maryland, 1982). Noise emissions from most power plants contain discrete tones, which can be more disturbing than broadband noise, but with one exception, these were not strong enough to increase annoyance potential in a series of measurements in Maryland (Maryland, 1982).

Coal Storage Piles. Coal power plants generally maintain ~90 days of coal in storage, 600-1800 m³ per MW capacity depending on the quality of the coal. Piles are typically 8-12 m high. Coal storage piles pose potential environmental problems of runoff, leachate, dust blow-off and, in case of spontaneous ignition, emissions of H₂S and other products of incomplete combustion. Little quantitative information is available on the latter two. Runoff is generally acid due to oxidation of sulfides in the coal. Quantities depend on rainfall and the area of the pile. Reported chemical and physical properties vary widely. Analyses of runoff water quality at two TVA plants are summarized in Table 2. Metals present in greatest quantity were copper, iron, aluminum, and nickel.

Table 2
Mean Values of Coal Pile Runoff

Characteristic	Plant	
	J	E
pH	2.8	2.7
Acidity (mg/l CaCO)	1360	3400
Sulfate (mg/l)	2780	5160
Dissolved Solids (mg/l)	3600	7900
Suspended Solids (mg/l)	190	470

Data of D. B. Cox and R.J. Ruane, in EPA, 1980.

Davis and Boegly (1981) report controlled, laboratory-scale leaching experiments. Type of coal was the most important parameter in determining leachate quality. Coal size and storage technique also affected quality, but their effects depended on coal type. Acidic, high sulfur coals of small particle size (i.e., high surface area) produced leachate of lower pH and higher dissolved metals content.

Solid Waste Disposal.* Character of coal power plant solid wastes depends on composition of the coal, processes used, air and water pollution control requirements, plant design, and operation procedures. A 1000-MW plant burning coal with 3% sulfur and 12% ash produces ~300,000 MT/y ash and ~600,000 MT/y wet scrubber sludge (with nonregenerable limestone flue gas desulfurization). In addition, solid wastes are produced by water treatment processes for make-up water, boiler blow-down, and coal pile runoff, as well as sanitary wastes from the >400 people working in a large power plant. Potential environmental problems depend on disposal methods, topography,

*Much of this section draws on work done with M.A. Crowther (Crowther and Morris, 1981).

geology, and climate. Because of the large quantities, solid waste disposal sites are maintained on or near the plant site if possible.

The combustion of coal produces an ash residue consisting of the inorganic mineral constituents present in coal, as well as the organic matter which is not fully burned. Two types of ash are produced during combustion: fly ash and bottom ash. Fly ash consists of the fine particulates that are entrained in the flue gas stream. Bottom ash is the coarser, heavier residue which accumulates in the furnace bottom either as a loose dry ash or as a slag.

The distribution of ash between bottom and fly depends on the type of boiler. In cyclonic units, the melting point of the ash is exceeded and therefore 80 to 85% of the ash is melted and collected as slag. The ash produced by pulverized coal units (which are the ones mainly used by utilities) is usually distributed as 65 to 80% fly ash and the remainder as bottom ash.

Coal ash consists of particles varying in size from <1 micron to 4 cm in diameter. The fly ash fraction generally consists of fine spherical particulates ranging up to 100 microns in diameter. As much as 5% by weight or 20% by volume in fly ash consists of cenospheres, which are silicate glass spheres filled with nitrogen and carbon dioxide. These very lightweight particles tend to float on ash pond surfaces.

The bottom ash fraction of the coal combustion residue is composed primarily of coarser, heavier particles with porous surfaces. If the bottom ash melts to slag at burner operating temperatures, the residue resolidifies as angular black particles with a glassy appearance.

The type of particulate collector employed determines the particle size distribution and total surface area of the collected fly ash. An electrostatic precipitator collects a much higher percentage of the very small particles (<1.5 microns) than does a mechanical collector. The collected fly ash, however, is much less sensitive to changes in efficiency of collection than is the fly ash released to the atmosphere. The difference is most dramatic in the total mass. A change from 98% removal of total mass to 99% removal results in slightly more than a 1% increase in mass of fly ash collected but a 50% decrease in the mass emitted to the atmosphere.

Mather (1961) analyzed size-differentiated fractions of fly ash and concluded that lithophile materials (alumino-silicates) and alkalis (Na and K) were generally more concentrated in the finer fractions and thus in ash collected by an electrostatic precipitator. Magnetite-hematite materials (iron bearing) were more concentrated in the coarser fractions of ash collected from the mechanical collector than in ash collected from the electrostatic precipitator. Carbon particles also increased in abundance as the particle size increased. Natusch et al. (1974) demonstrated that toxic trace elements, arsenic, antimony, cadmium, lead, selenium, and thallium, were most concentrated in the smallest respirable particles emitted from coal-burning plants. These elements are probably volatilized during combustion and preferentially adsorb or condense on small particles which have the greatest surface area. As collection efficiency is increased, larger fractions of these particles with high concentrations of toxic elements will be collected as solid waste. Since the total additional mass is small, however, this will not lead to great changes in the overall concentration of these toxic elements in the solid waste.

The chemical makeup of coal ash depends largely on geologic and geographic factors related to the origin of the coal. Many elements are enriched in coal in comparison to their abundance in the earth's crust. When the coal is burned, some of these elements are further enriched in the ash residue. As a result, some potentially toxic heavy metals exist in coal ash in concentrations much greater than found in either the input coal or the earth's crust.

The major constituents of coal ash, primarily silicon, aluminum, iron, and calcium, make up 95 to 99% of the total composition. Minor constituents, such as magnesium, titanium, sodium, potassium, sulfur, and phosphorous, generally comprise 0.5 to 3.5%. Table 3 gives the range of variation of the principal constituents of coal ash.

Coal ash also contains a large number of trace elements, and a number of studies have attempted to quantify them. Several recent studies (Cowherd et al., 1975; Kaakinen et al., 1975; Klein et al., 1975; Radian, 1975) indicate that the elements present in coal ash are distributed into the

Table 3
Major Chemical Constituents of Coal Ash

Constituents	Range (%)
Silica (SiO ₂)	6.0 - 58
Alumina (Al ₂ O ₃)	4.0 - 44
Ferric Oxide (Fe ₂ O ₃)	1.0 - 44
Calcium Oxide (CaO)	0.2 - 52
Magnesium Oxide (MgO)	0.1 - 14
Titanium Dioxide (TiO ₂)	0.4 - 4.17
Potassium Oxide (K ₂ O)	0.1 - 4.0
Sodium Oxide (Na ₂ O)	0.09 - 28
Sulfur Trioxide (SO ₃)	0.1 - 32
Carbon (C) and Volatiles	0.1 - 20

Source: Fred C. Hart Associates (1978).

fractions of coal combustion residue (bottom ash, fly ash, and vapor) according to definite patterns based on volatilization temperatures. Klein et al. (1975) indicate that the elements appear to divide into three main classes, as follows:

1. Elements that are not volatilized in the combustion zone, but instead form a rather uniform melt that becomes both fly ash and slag. These elements include Al, Ba, Ca, Ce, Co, Eu, Fe, Hf, K, La, Mg, Mn, Rb, Sc, Si, Sm, Sr, Ta, Th, and Ti.
2. Elements that are volatilized on combustion and condense or adsorb on the fly ash as the flue gas cools, leading to depletion from slag and bottom ash and concentration in the fly ash. These elements include As, Cd, Cu, Ga, Mo, Pb, Sb, Se, and Zn.
3. Elements that remain almost completely in the gas phase and are discharged to the environment as gases. These elements include Hg, Cl, and Br.

The elements Cr, Cs, Na, Ni, U, and V were not placed into one of these three groups but were judged to exhibit behavior intermediate between the first two groups. Most sulfur is emitted as SO_x , and the halogens as hydrogen halides, all of which are scrubbed in an alkaline SO_x scrubber (CaO , $CaCO_3$, or $NaSO_3$). The conclusions of the other studies cited are generally consistent with the Klein results, which is notable considering the differences in the furnace types, coal types, and sampling and analytical procedures used.

Oak Ridge National Laboratory has analyzed coal ash for hydrocarbon content (Van Hook, 1976). Individual hydrocarbon compounds exhibited concentration ranges from 66 to 816 ppb, with a total detected concentration

of about 9 ppm. The C28-C30 hydrocarbons were in greatest concentration, with C29 the highest. After a florisil and alumina column chromatograph cleanup, the same ash was analyzed for the presence of polycyclic aromatic hydrocarbons (PAH). Concentrations were low, ranging generally from about 10 to 20 ppb. The total PAH concentration was estimated to be a maximum of about 0.2 ppm.

Recent studies (Coles et al., 1978; Styron et al., 1979) have looked at the fate of natural radionuclides in coal, ^{238}U , ^{235}U , ^{228}Tn , ^{228}Ra , ^{226}Ra , and ^{210}Pb . All of the radionuclides studied are enriched in ash relative to the input coal and in general tend to concentrate in the finer particles. While there are some lignites that have relatively high radionuclide content, these are not being used and need not be used as fuel.

A number of reports have been published on the mutagenicity of coal fly ash (Chrisp et al., 1977; Fisher et al., 1979; Kubitschek and Venta, 1979). Reports from two different groups sampling at seven power plants have concluded that although fly ash leaving the stack to the atmosphere is mutagenic in the Ames Salmonella reversion assay, the ash that is collected as solid waste in the electrostatic precipitator (ESP) is not mutagenic in this assay.

Bonnell and Schilling (1978) also reported no mutagenic activity in fly ash collected in the ESP hoppers of a large British coal-fired power plant. Fisher et al. (1979) conclude that if there is any mutagenic activity in the collected ash, it must be less than 0.2% of that in the material emitted to the air. Apparently the mutagenic activity is due to compounds volatilized during combustion and adsorbed on the surface of fine particles, which present the most surface area, as the effluent stream cools. This

presumably takes place after the exhaust gases pass through the precipitator. Fisher et al. (1979) draw the conclusion that use of low-temperature precipitators could lead to contamination of the bulk of the collected fly ash with mutagens. They recommend high temperature collection of particles followed by cooling of the effluent stream and separate collection of the condensed PAH. The latter would yield a hazardous solid waste that is much smaller in volume. It is not yet clear whether the risk from mutagenic fly ash in the air warrants this step, or even if the overall level of risk would be improved. It is also not clear to what extent this already happens in a scrubber. In addition to the mutagen assay studies, there have been reports of specific mutagenic agents in fly ash. There is considerable controversy over these findings, and their applicability to collected fly ash is unclear. Dow Chemical Corporation reported 38 ng/g tetrachlorodibenzo-p-dioxin (TCDD) in the stack effluent of an industrial power plant burning coal and fuel oil (Dow, 1978). These findings are disputed in a report finding no evidence of TCDD in the fly ash emitted from a coal-fired power plant stack (Kimble and Gross, 1980). No measurements have apparently been made for TCDD or other dioxins in fly ash collected as solid waste. There is every reason to believe, however, that the solid waste would have lower concentrations of dioxins than the emitted fly ash.

Lee et al. (1980) reported finding dimethyl and monomethyl sulfate in the particles emitted from a soaker-type coal-fired heating plant and a large coal-fired power plant. As with TCDD, one would expect lesser quantities in the collected ash. Moreover, there is some question as to the general applicability of the results.

These studies would indicate that there is no risk of mutagenic activity in fly ash collected in current ESP hoppers of large coal-fired utilities. Because of the limited number of plants studied, however, this conclusions cannot be drawn generally with much confidence.

Ash deposited in lagoons may pose a dust hazard as the material dries out. The small, hollow cenospheres have a low density and are susceptible to wind erosion. Proper compaction of the ash helps to prevent this. Also, in some instances the ash tends to form a skin that, if undisturbed, protects it against blowing. Spraying the surface with polymer emulsions has also proved effective in preventing wind erosion. Long-term control is best achieved by establishing a suitable vegetative cover.

Ash does not readily support plant growth. This is due partly to lack of the necessary nutrients (a need that can be supplied by application of fertilizers) and partly to the presence of inhibiting elements such as boron. Yet, natural colonization of an ash surface by vegetation often takes place. The first plant to appear is usually a moss, which can cover a moist ash surface within six months and which effectively binds together the ash particles to inhibit erosion. A common weed is usually the next colonizer to appear, and this can be followed by a range of other plants (DOI, 1974).

Tests on ash piles have shown that boron may be 20 times as available in fresh ash as in a normal soil. Chemical treatments for removing the excess boron have proven to be too expensive. Fortunately, various useful plant species tolerate ash conditions. The clover family is, in general, quite tolerant, as are some grasses, particularly rye grasses. Of the arable crops, the beet and cabbage families do well. Sugar beets, fodder beets, and mangels benefit from the boron in the ash, provided the content

is not too high. Considerable success has been obtained with rye and wheat on a field scale, and (at a later stage) even with the relatively intolerant barley (DOI, 1974). Of course, if the ash can be covered with topsoil to a depth of perhaps 12 inches, then a great range of crops can be grown on the site. Cost usually makes this impractical.

A variety of trees (alder, honey locust, spruce, poplar, and willow) are tolerant to ash conditions, but their growth is not sufficiently vigorous to sustain commercial timber production. A number of shrubs can also be grown on ash. Thus it should be possible to landscape an ash disposal site with these tolerant species.

Fly ash is used as a building material in concrete, brick, and structured fill. Such use is more common in Europe than in the United States. The typical method of ash disposal is sluicing to settling ponds, followed by removal and disposal as fill (Theis and Marley, 1979). Water requirements for sluicing vary widely in the United States, but exceed 38,000 L/d per MW capacity at 40% of coal fired power plants (Kim, Chu, and Ruane, 1983). Surface water can become contaminated by runoff from disposal piles and groundwater by leaching of materials from the waste. While concentration of trace pollutants in the ash can be estimated (Crowther, Thode, and Morris, 1980), concentrations in leachate are not usually predictable without testing. Testing procedures have been developed by ASTM and EPA. A number of studies have reported data on quality of leachate and ash pond overflow (ANL, 1976; Theis and Marley, 1979; Kessler, 1981; EPA, 1980). Release depends upon concentration of each trace contaminant in the ash, the amount and type of ash in the pond and the manner of loading, the chemistry of the contaminant, and the pH which is itself dependent on the ash.

Interest has recently centered on radioactive releases from ash piles, particularly in comparison with uranium mill tailings. A careful balance made in one power plant found over 90% of the uranium-234 and -238 and lead-210 and 50 to 75% of the polonium-210 in the coal are collected as solid waste in the ash (Styron et al., 1979). Some studies have indicated that this could be a problem but they were based on coals with exceptionally high uranium content. Such coals represent only a small fraction of U.S. reserves and are not at present mined (Teknekron, 1980). Beck et al. (1978) conducted a detailed and thorough analysis of the radiological hazard associated with coal and ash. On the basis of more valid data concerning uranium concentrations in coal, particularly new empirical data concerning the radon emanation rate of coal ash (about one-twentieth the rate of uranium mill tailings), this study concluded that:

The possibility of a gradual increase in atmospheric ^{222}Rn levels due to the buildup of ^{238}U and its daughters in waste ash disposal ponds seems small, since it appears that the emanation of radon from ash is considerably lower than from other materials such as soil or uranium mill tailing.

Using USGS data on coal characteristics (Swanson et al., 1976) and emanation rates from Beck et al. (1978), Nagy (1980) compared ^{222}Rn emanation from various coal and lignite ash piles and uranium mill tailings, finding radon emission potential of uranium tailings piles to be 300 to 2000 times that of coal ash.

Scrubber sludge is often mixed with ash for disposal. Flue gas desulfurization (FGD) sludge varies greatly in composition depending on coal composition, the type of scrubber used, the degree of fly ash removed prior

to scrubbing, operating conditions and the type and amount of additives. Sludge from a lime/limestone scrubber, most frequently used by utilities, is mainly calcium sulfite, calcium sulfate, calcium carbonate, and ash.

Concentrations of some components in scrubber liquors can exceed water quality criteria by more than an order of magnitude. These substances can be introduced into water supplies by overflow from the disposal pond or by percolation through the settled solids into the soil (ANL, 1976). Contamination of groundwater supplies by leaching is a definite threat because of the permeability of the wastes. Untreated wastes, with permeabilities in the range 10^{-4} to 10^{-5} cm/sec (10 to 100 ft/yr), offer little resistance to flow of scrubber liquor to the pond floor.

During the active life of the pond, while sludge is continually being deposited and there is supernatant scrubber liquor, the concentrations of leachate components will approach their concentrations in steady state scrubber liquor. If the wastes are still physically unstable and cannot be protected from exposure to rainfall when operations cease, leaching will continue. Although the concentrations of trace sludge components in the leachates will gradually decrease, those of major components will remain constant. The possibility therefore exists for infiltration of substantial quantities of bulk and trace components of scrubber liquors into the underlying soil.

The rate of migration from the pond of trace elements in the leachates can be slowed by the retentive characteristics of some soils, but it is known that some substances, such as boron and selenium, are less effectively attenuated by soil than others. Laboratory attenuation (percolation) data obtained in soils at utility stations indicate that after 10 years of

continuous flow, leachate concentrations were diminished by factors of 95% or more for most soils, a reduction more substantial than that for other contaminants such as sulfate, total dissolved solids (TDS) and hardness, which remained undesirably high. Certain trace elements -- specifically barium, boron, chromium, and selenium -- also remained high, exceeding ten times the drinking water standard (Fred C. Hart, 1978).

Adverse effects on the quality of surface and ground water have not been detected thus far, but the limited time and extent of investigation provide little assurance.

A possible source of air quality effect is wind-whipped spray from the surface of sludge impoundments. While no quantitative estimates of this problem were found, Aerospace Corporation (1978) speculated that a buffer zone may be required for sludge impoundments. Initial measurements on FGD sludge ponds have shown emissions of volatile gases. These were largely reduced sulfur gases: hydrogen sulfide, carbonyl sulfide, dimethyl sulfide, carbon disulfide. Emission rates were reported to be approximately 2 kg/day (as sulfur) from a 100-acre pond. In addition, traces of benzene, toluene, and pinene were identified in the emissions (Adams, 1979). Mechanisms and controlling factors of chemical volatilization from waste sites are discussed in detail by Shen and Tofflemire (1980). Processes are (1) volatilization of organic liquid chemical; (2) volatilization of liquid chemicals from water, and (3) volatilization of chemicals adsorbed to particles.

Disposal ponds used for untreated FGD sludge may be reclaimed by allowing the sludge to dry and covering it with a soil overburden, but use of the land may be limited. Proper contouring to control rainfall runoff

and minimize percolation of water through the overburden will be necessary to avoid resaturating the sludge.

If the sludge has been stabilized prior to disposal, the reclaimed land will have greater stability and should be suitable for wildlife, recreation, agriculture, or light construction. The stabilized sludge would not provide sufficient strength for heavy construction, and ground water at or near the site would have to be used with care (Teknekron, 1980).

Cooling Water. About 85% of the waste heat generated in a coal power plant is taken up by water coolant (over 4000 Btu/kWh_e). Potential environmental impacts related to cooling water needs of coal power plants are well known and are listed in Table 4. These problems will be covered only briefly.

Table 4
Environmental Impacts of Power Plant Cooling Systems on Water Bodies

- Entrapment and entrainment of fish and other aquatic life in the intakes
 - Evaporation loss of water
 - Effect of calcification due to heated discharges on aquatic life -- and of cold shock during plant shut-downs
 - Chemical pollution of receiving water bodies from biocides, and corrosion and scale inhibitors
 - Noise
 - Microclimatic affects
 - Particle release from cooling towers
-

Impingement can have a greater effect on fish than thermal discharges. Grotbeck and Bechthold (1975) estimated 3000 fish killed due to impingement June through September on a plant on the Mississippi River.

Using an integrated community approach examining both long- and short-term variation, Cole (1978) was able to determine subtle effects of

entrainment in a power plant on Lake Erie with once-through cooling. Mean primary productivity and zooplankton density declined ~50% as water passed through the cooling system. Productivity recovered as the water passed out of the plant and mixed again with lake waters. Plankton loss was estimated to have a negligible effect because of the large volume of water in the lake. Community respiration nearly doubled in the cooling system, presumably due to decomposition of dissolved organics. Respiration recovered to normal upon remixing in the lake. Mortality of entrained fish larvae was high (70%). Less than 1% of most fish populations were entrained, but 5-15% of some species may have been entrained. What fraction of larvae death would eventually affect species population is unknown. Estimates as low as 5% have been suggested but are controversial.

Thermal discharges, leading to califaction of receiving waters, can cause diminished oxygen capacity, increased oxygen demand, and increased toxicity to biocides. Sufficiently increased temperature can cause death. The difference between optimal and lethal temperatures varies by species, but can be as small as 4°C. More important, the temperature range for breeding is much narrower (Hargis and Warinner, 1969).

Aquatic life which becomes acclimatized to the warmer water may die during a winter plant shut-down.

Roessler and Tabb (1974) studied the effect of a 35,000 l/s heated discharge from 2 fossil power plants on Biscayne Bay, a shallow 56 km x 12 km bay in Florida. The operation of the plant had previously been shown to have influenced the chemistry of the bay only in the immediate vicinity of the discharge and no significant chemical-induced alteration had occurred. It was found that an area of about 300,000 m² had average temperature

elevations above 4°C, 400,000 m² between 3 and 4°C, and 500,000 m² between 2 and 3°C. A total of ~1,200,000 m² showed a decline in abundance for at least part of the year, although in ~40% of this area increased winter catch compensated for decreased summer catch. It was estimated that in 16% of the area, recovery and recolonization would be slow were the discharge stopped because of the death of vegetation (*Thalassia*) and changes in the sediment.

Cooling water is treated with biocides to prevent slime formation which interferes with heat exchange. Chlorination is the most common method. Chlorine reacts with other water constituents to form chlorinated organics in part per billion (ppb) concentrations which are of environmental and health concern (EPA, 1980). Helz et al. (1984) traced colloidal bromocarbons in the discharge plume of a 1300-MW coal-fired plant over 6 km. Only traces of trihalomethanes were observed, however, presumably as a result of rapid loss of chlorine in the power plant.

Recirculating cooling water systems reduce water intake requirements and discharge quantity. Cooling ponds are used in dry climates and in areas where large land areas are available. Elsewhere, mechanical or natural draft evaporative cooling towers are used. Evaporation leads to an increase in dissolved solids in the water. A portion of the cooling water called "blow-down" is discharged to avoid this. Blow-down water thus has a higher solids content than intake water. In addition, it contains biocides, scale and corrosion inhibitors, and possibly asbestos from asbestos-cement natural draft cooling towers. A survey of 82 U.S. coal plants found a mean blow-down rate of 11,000 l/d discharged per MW capacity (EPA, 1980).

In addition to discharges to water, cooling towers discharge to air. Meteorological effects of cooling towers are listed in Table 5. Calculated

drift from natural draft cooling towers varies from 0.01 to 0.3% of recirculating water rate. Drift from a 2500 MW nuclear plant was calculated to contain over 3 MT of solids per day (AEC, 1972). When brackish water is used for cooling, salt deposition from the cooling tower plume may be of concern. Concern has also arisen over the distribution of chemicals and microorganisms through cooling towers when cooling water is taken from highly polluted sources. Finally, mixing of cooling and combustion gas stack plumes can lead to increased heterogeneous oxidation of sulfur dioxide to sulfate (Meagher et al., 1982).

No measurable increases in airborne salt concentrations at monitors 160 to 1700 m from the cooling towers resulted from the operation of mechanical cooling towers at the Turkey Point Plant (Hunter, 1976). Studies at Chalk Point 3 which has approximately equal emission rates of brackish water from its cooling tower and stack scrubber found maximum deposition within 1 km of the source with a measured deposition rate of 8 kg/ha-month with no apparent build-up of sodium ion in the soil (Maryland, 1982). Studies on the effects of saline aerosols from cooling towers have shown that stage of development, species, and phenotype determine susceptibility of the plants to this impact. Median effective doses varied more than 100-fold among species, from 2.4 to 264 $\mu\text{g Cl/cm}^2$ (McCune et al., 1977).

Table 5
Effects of Cooling Towers on the Atmosphere

- Local ground level fogging
 - Icing of nearby roads and structures
 - Deposition of chemically laden drift particles, especially when salt water or chemically or biologically contaminated water is used for cooling
 - Consumptive use of water by evaporative loss
 - Micro climate modifications (wind, precipitation, cloud cover)
 - Acid rainout (H_2SO_4) due to comingling of cooling tower vapor plumes with stack gas plumes
-

After Shirazi et al., 1977.

4. REGIONAL EFFECT -- AIR POLLUTION

Early coal power plants with little pollution control and short (~50 m) stacks had a substantial impact on the immediately surrounding area in air pollution and soot fallout, affecting health, vegetation, and materials. Movement to larger plants with more efficient combustion and >98% particle removal and sulfur control and tall (300 m) stacks reduced local peak pollutant levels and made air pollution more of a regional than a local concern. Table 6 lists the principle air pollution emissions from a modern U.S. coal fired power plant. Particles are generally small, having escaped through electrostatic precipitators and possibly scrubbers or other control devices. They are composed of inert materials (e.g., ferro silicates) sulfates, trace metals, and organics. Upon leaving the plant, the pollutants are subject to atmospheric chemical reactions, many of which are photochemical, leading to formation of secondary sulfate and nitrate particles, and ozone. They can impact areas hundreds of kilometers from their origin and are eventually removed by both wet and dry deposition. Radionuclide releases are discussed below. Principle effects of concern are listed in Table 7.

Table 6
Coal Power Plant Air Emissions per GW_e-y

	Metric tons
Particles	1.5x10 ³
SO ₂	2.9x10 ⁴
NO _x	2.9x10 ⁴
CO	2.1x10 ³
CO ₂	1x10 ⁷

From DOE, 1983.

Table 7
Air Pollution Effects

Effects	Air Pollution Agents
Human health effects	Gases, fine particles, acid fog
Agricultural crop damage	Gases, wet and dry deposition
Effects on natural terrestrial ecosystems	Gases, wet and dry deposition
Effects on lakes, soils	Wet and dry deposition
Visibility	Fine particles
Materials	Gases, acid fog

Health Effects. Health effects of air pollution from coal combustion include acute effects such as increased attacks in asthmatics, increased respiratory discomfort in bronchitics, and increased acute respiratory disease in children. These acute effects of short (1-3 day) exposures are well documented (Goldstein, 1983; Douglas and Waller, 1966). Acute effects on respiratory function have been demonstrated in clinical experiments (Hackney et al., 1979). Long term chronic effects of exposure to current levels of pollution have also been shown (Detels et al., 1981), but are more controversial. Excessive acute respiratory disease in childhood may increase the likelihood of chronic disease in later life (Burrows et al., 1977).

For several years, many health risk assessments have used a sulfate surrogate model developed by Morgan et al. (1978) to estimate health effects of air pollution in terms of increased mortality rates. At the time of its development, animal toxicology (Amdur, 1976), epidemiology (Environmental Protection Agency, 1974), and correlational analysis (Lave and Seskin, 1978) all seemed to point to sulfate as the harmful agent in the sulfur-particulate mix of air pollution. Even at that time, however, the Morgan et al. assessment of the evidence was that, if the usual statistical

requirements of "proof" in science were applied, the conclusion was "not proved." For questions of public policy, "not proved" does not mean the best estimate is zero. On the contrary, adverse effect on health from sulfate exposure seemed likely.

A recently-completed study prepared for the National Science Foundation (Morgan et al., 1982) reevaluated the question of the sulfate health effects damage function by eliciting quantitative best judgment estimates from a number of air pollution health effects experts. There is a wide range of expert opinion in this area. Some experts believe that there are no health effects, and certainly no mortality effects, associated with sulfate exposure near current ambient levels. Other experts believe this is likely to be the case but will not rule out some possibility that there might be effects which may even be quite large. Yet other experts believe that while there is some possibility of no effects, the likelihood is that there are health effects of sulfate even at current ambient levels.

Quantitatively, the NSF study found a range of dose-response estimates roughly similar to the range in the earlier sulfate damage function (0 to 12 deaths per hundred thousand person-microgram sulfate per cubic meter). The study concluded that the boundaries of this range represented quite different schools of thought and that describing the range of possible effects in terms of a continuous distribution was inappropriate.

A difficulty with any surrogate is that it can be mistaken for the true (but unknown) causative agent. Use of a sulfate damage function could make it appear that merely reducing sulfur dioxide emissions would improve health. But a surrogate function cannot accurately predict the effect of reductions in a single pollutant: health impacts depend on exposure to the

total mix. Moreover, use of the sulfate surrogate has tended to shift emphasis away from effects of other pollutants that may not be well-represented by sulfate. Selection of a surrogate also has more subtle effects in the analysis itself. Because of the time of chemical reactions in sulfate formation, numbers and places of people exposed may differ from those were fine particles the surrogate.

Some epidemiology now suggests that fine particles may be a better surrogate for estimating air pollution health effects. Indeed, the Harvard University group (Ozkaynak et al., 1983) now proposes a fine-particle health damage function of 1.3 annual deaths per 10^5 person- $\mu\text{g}/\text{m}^3$. For the time being, we believe the sulfate damage function can serve as a reasonable way of estimating health effects of the sulfur-particle air pollution mix (as does the Harvard University group in the absence of measurement data on fine particles). There are not an abundance of data on fine particles, especially measurements of actual exposure. Sulfates make up a substantial fraction of fine particles and are better correlated with fine particles than are the more readily-available measures of total particles.

Oxidant pollutants, such as NO_2 and O_3 , have effects quite different from the reducing sulfur particulate mix and are not well correlated with sulfate. While a number of clinical and toxicological studies link NO_2 with health effects, epidemiological research -- principally based on indoor exposures -- yields mixed results, and a sufficient body of knowledge on which to base a dose-response function suitable for assessment is only beginning to develop (Novak et al., 1983).

Fischer et al. (1983) developed an upper bound of potential sulfate effects from Morgan et al. (1982) and expanded the scope of analysis by

developing an upper bound of potential NO₂ effects on mortality. Their estimates of upper bound air pollution effects from the coal electric fuel cycle in the U.S. (mostly due to power plant emissions) are given in Table 8.

Table 8
Upper Bound Air Pollution Health Effects Estimates
Mortality Risk per GW_e-y

Pollution	Mortality Risk
Sulfate	0 - 120
NO ₂	0 - 120

Derived from Fischer et al. (1983).

Although the median NO₂ dose-response coefficient was only one-tenth that of sulfate, differences in population exposure and greater uncertainties in both exposure and dose-response for NO₂ led to similar upper bound effects. These estimates are based on modeled typical population exposure in the U.S. and, of course, vary by plant location, population distribution, and meteorological patterns.

Radionuclides in Coal.* Coal contains trace amounts of three naturally occurring radioactive decay series headed by uranium-238, uranium-235, and thorium-232. These radionuclides are present in the particles emitted during coal combustion. Although particle collection devices may control such emissions, radon gas in the coal will escape to the atmosphere even if particles are completely collected.

*This section draws on work by J. Nagy (Morris et al., 1980).

Radon gas is essentially trapped in the coal in secular equilibrium with the parent radionuclide. When coal is burned, the confining matrix is destroyed and radon gas then existing is released to the atmosphere. The risk associated with radon, however, is considered small compared with that from radionuclides in particulate emissions (EPA, 1979).

It has been suggested that in the future the coal fuel cycle will involve the same amount of uranium per unit energy as the nuclear fuel cycle. This thesis is based on the very high uranium content of a few lignite beds in the western United States. A typical coal, however, contains about 2 ppm uranium and 4 ppm thorium (Swanson et al., 1976). On a unit energy basis, therefore, the uranium/thorium inventory in the coal cycle is orders of magnitude less than in the nuclear fuel cycle. Maximum dose to individuals in the vicinity of a coal-burning facility may, however, be comparable to maximum dose from a nuclear facility; and dose to bone may be higher owing to the relative isotopic constituents of the two fuels (McBride et al., 1977; EPA, 1979).

Solubility of the particulate matter influences both the amount of radionuclide taken up by green plants and the amount of inhaled or ingested fly ash cleared from the human lung or digestive tract. In both cases, solubility -- and therefore potential impact -- is believed to be low (Beck et al., 1978; EPA, 1979).

There are many human exposure pathways for radionuclides. Individuals are exposed to low-LET gamma radiation externally by immersion in an atmosphere containing suspended (or re-suspended) radioactive particles and to particles deposited on the ground. Individuals are exposed internally via inhalation or ingestion. Sources of ingested material include foliar

deposition and radionuclides extracted from the soil by plants. Internal exposure involves high-LET alpha radiation as well as beta and gamma radiation.

Beck et al. (1978) surveyed the area around three coal-burning plants and could find elevated levels of radionuclides at only one, which had concentrations comparable to what their model predicted. They report a projected human health impact only via the inhalation pathway and only for the maximally exposed individual.

Table 9
Latent Cancer Mortality from Coal Radionuclides*
(Excess Deaths per GW-y)

	"Old" Plant	"New" Plant
EPA - Site A	6.9	2.9×10^{-1}
- Site B	4.6×10^{-1}	2.4×10^{-2}
- Site D	1.4×10^{-2}	7.0×10^{-3}
- Site F	2.0×10^{-3}	1.0×10^{-4}
McBride et al.	--	2.0×10^{-2}

Comparison of Parameters

	EPA "Old"	EPA "New"	McBride et al.
Net particle control (%)	84.0	99.2	99.0
Stack height (m)	97	185	100
U (ppm)	1.9	1.9	1.0
Th (ppm)	5.0	5.0	2.0
Sites	A: Urban B: Suburban D: Rural F: Remote	A: Urban B: Suburban D: Rural F: Remote	Midwestern
Region (km)	80	80	88.5
Population (10^6)	A: 17.1 B: 2.4 D: 0.6 F: 0.001	A: 17.1 B: 2.4 D: 0.6 F: 0.001	3.5
Food produced in area (%)	92-100	92-100	100
Enhancement (enrichment)	All: 1.0	U: 2.0 Ra: 1.5 Pb: 5.0 Po: 5.0	All: 1.0
Particle solubility	Insoluble	Insoluble	Soluble

*Source: McBride et al. (1977); EPA (1979).

McBride et al. (1977) and EPA (1979) modeled health effects as a function of a number of variables including location, stack height, and consumption of locally produced food. Table 9 shows projected health impacts as latent cancer mortality. The EPA dose-response values have been applied to the McBride et al. dose results and impacts normalized to one GW-y of electric energy production.

Acid Rain. Acid rain, including dry deposition and snow melt, clearly affect unbuffered lakes in New York State, New England, Canada, and Scandinavia. Effects on more well-buffered areas, or agricultural crops, natural terrestrial ecosystems, and human health (e.g., via leaching of toxic trace metals into water supplies and aquatic food chains) are slower acting and more controversial.

Unpolluted rain is mildly acidic, with a pH of 5.6 due to carbonic acid formed by CO₂ in the atmosphere. Acid precipitation -- with pH of individual storms ranging as low as 2.4 -- generally contains about 60 to 70% of its acidity in the form of sulfate ion and the remainder as oxides of nitrogen (Likens, 1976). In the U.S., roughly half the nitrogen oxide emissions come from stationary sources such as industry and power plants; the rest is attributable to transportation. Most sulfur oxide emissions, however, originate in stationary coal- and oil-burning facilities (EPA, 1978).

Precipitation in the eastern United States has become increasingly acid, and rainfall pH in this region is now uniformly below the value of 4.6 sometimes quoted as characteristic of acute effects. Moreover, dry deposition or direct absorption of atmospheric gases is also of significance.

The principal variable determining the aquatic impact of "acid rain" -- a term commonly encompassing both wet and dry deposition -- is the natural soil alkalinity of the watersheds tributary to fresh water bodies. Carbonate rock, usually limestone, provides in many areas an alkaline chemical buffering mechanism that largely neutralizes the tendency of acids to lower aqueous pH. Areas with siliceous bedrock (cf. Likens et al., 1979) or thin soils, in contrast, have little or no buffering capacity and are more sensitive to acidification. Such regions as the Adirondack Massif, the Scandinavian peninsula, and Canada's Laurentian Shield fall into this category; it is here that the most pronounced effects on fresh waters have been observed. Nevertheless, the buffering capability even of alkaline soils may eventually be exhausted, allowing sudden drops in pH. There is some evidence that this exhaustion is indeed occurring (Hendrey, 1980).

The effect of lowered pH on lakes and streams is complex, but there are certain regularities (Hendrey et al., 1980). In general, there is an accelerated aging of the fresh water system toward a more sterile, oligotrophic state. Sensitive species of plants and animals disappear and are replaced by acid-tolerant species, while diversity of the remaining species is sharply reduced.

Synergistic effects begin to occur: fish experience not only direct stress, thought to result from disturbances of the cellular salt balance and probably also from toxic effects of acid-soluble metal ions, especially aluminum (Schofield, 1978), but also the reduction of natural food sources as sensitive invertebrates die off.

During spring spawning, melting snow delivers an "acid pulse" of accumulated deposition (Schofield and Trojnar, 1979) that has been

tentatively linked with reproductive failures of fish populations. It is not known whether such episodic extremes, which also can occur during rainfall events, are more or less detrimental than continuous exposure to milder background acidity; both effects are probably important.

Eventually, it becomes impossible for the fish population to reproduce. Paradoxically, in these final stages of acidification the remaining adult fish often grow to record size because of a decline in competition; and the water is unusually clear because of an absence of most forms of phytoplankton and zooplankton. Thus a dying lake can be aesthetically and recreationally attractive for a time.

Terrestrial plants response to acid rain varies greatly by species, environmental setting, and exposure. Acid rain can produce lesions on leaves and reproductive structures leading to effects on respiration and photosynthesis. Experiments with agricultural crops have shown significant reductions in overall productivity (Evans et al., 1980).

Cross-sectional studies of soybean yield have shown productivity decreases associated with acid rain, but the effect was less than that of ozone (Medeiros et al., 1984). Productivity effects in natural forests are more difficult to measure, since the effect takes place more slowly and there are many more uncontrolled covariates. There have thus been conflicting reports (Tamm, 1976; Jonsson, 1977; Abrahamson et al., 1977; Cogbill, 1977).

Long-term effects on natural terrestrial ecosystems may be associated with changes in soil chemistry and microbial populations. Decomposition rates of organic matter, nitrogen transformation, and ultimately overall nutrient cycling may be affected.

3. GLOBAL EFFECTS*

The earth's temperature is determined in large part by thermal balance between net incoming solar energy and heat radiated from earth's surface into space at longer wavelengths. Water vapor, ozone, carbon dioxide, and other polyatomic molecules naturally present in the atmosphere interfere with this heat rejection by capturing and radiating back some of the long wavelength radiation. Were it not for this atmospheric blanketing effect, global temperatures would be much colder than present (Zimen, 1979), with serious ramifications for terrestrial life.

Accelerating CO₂ production projected by virtually every scenario of world energy growth has caused substantial concern that during the next century increasing temperatures may result in melting of polar ice, rise in sea level, and geographical shifts in agricultural productivity.

Carbon dioxide production from fossil fuel combustion has been rising in a quasi-exponential fashion since about 1860, at a typical rate of ~4.3% per year (Keeling, 1973; Rotty, 1979). Carbon dioxide increases observed in the atmosphere have equalled about half the cumulative emissions from fossil fuel combustion (Siegenthaler and Oeschger, 1978), indicating that some part of the ecosphere is acting as a "sink," or repository for CO₂.

The global carbon budget is poorly understood; yet it is clear that if current relationships continue to hold, atmospheric CO₂ concentrations will continue to increase at a rate paralleling that of fossil fuel emissions. Coal produces more CO₂ per unit of useful energy than does petroleum or natural gas. Model projections indicate that atmospheric CO₂ levels could

*This section draws on a review done by H. Fischer (Morris et al., 1980; Fischer, 1980).

double from preindustrial levels by the third quarter of the next century (NAS, 1983).

More important than average global temperature is its spatial variation. It is nearly certain that warming would be greatest at the poles, particularly the North Pole, which has no stabilizing land mass. Polar warming from a single CO₂ doubling is estimated to be about 8°C, compared with perhaps 2°C at the equator.

While melting of floating ice would not affect ocean levels, rapid glacial surges might take place if warmer water destabilizes the landed ice in Antarctica, possibly leading to sea level increases from 5 to 30 meters in as little as 100 years (Mercer, 1978; Hollin, 1970).

North-south polar asymmetry would contribute to what may be the most significant effect of global warming -- a shift in atmospheric circulation patterns expected as a result of changes in the temperature gradient from warm to cooler regions. These patterns determine which areas of the world are deserts and which have high rainfall, and their seasonal movements govern such processes as the arrival of the monsoons. The extent of such movement is not currently known, but the paleoclimatological record gives evidence that during the last major warm period, from 8,000 to 4,000 years ago, North Africa was considerably wetter and the U.S. plains states were drier than today (Kellogg, 1977). It is also known that sea level changes have accompanied geological warm periods, although it appears that the Arctic ice pack has been a perennial feature for the last million years (Hunkins et al., 1971).

The degree of impact regional climatic shifts have on agriculture depends in part on whether suitable soils and climate are matched or the degree to which either soils or cultivars can be adapted.

The oceans are another food source that may be affected by changes in atmospheric patterns. The world's greatest fisheries are produced by upwelling of nutrient-rich waters driven by an interaction of persistent air circulation cells with coastline topography. If atmospheric patterns shift geographically, these major fisheries may be shifted as well; and their productivity may be increased or decreased, depending on topography and on the nutrient content of the waters involved. Although the impact is uncertain, the history of the Peruvian anchovy fishery (Idyll, 1973) indicates that a small shift in atmospheric circulation can have an immediate (<1 year) and substantial adverse impact on the fishery.

International socio-political changes would seem likely as a result of shifts in agricultural productivity, decline in global agricultural productivity during the shift, and concurrent effects of rising sea level on coastal cities.

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