

Appendixes



Groundwater Contamination and Its Impacts

- A.1 AN APPROACH TO ASSESSING THE HEALTH RISKS OF CHEMICALS IN GROUNDWATER (p. 243)
- A.2 SUMMARY OF TOXIC EFFECTS OF ORGANIC AND INORGANIC CHEMICALS KNOWN TO OCCUR IN GROUNDWATER (p. 248)
- A.3 FREQUENCY OF DETECTION OF SELECTED CHEMICALS IN GROUNDWATER (p. 262)
- A.4 SUBSTANCES IN GROUNDWATER WHOSE DETECTED CONCENTRATION HAS EXCEEDED STANDARDS AND TYPES OF STANDARDS EXCEEDED (p. 264)
- A.5 SOURCES OF GROUNDWATER CONTAMINATION (P. 267)
- APPENDIX A REFERENCES (p. 289)

A.1 AN APPROACH TO ASSESSING THE HEALTH RISKS OF CHEMICALS IN GROUNDWATER

Because of uncertainties about the relationship between exposure (e. g., to chemicals) and impacts on human health, public health efforts are based on identifying *probabilities* of impacts. This process entails identifying when exposure is likely to pose either *significant* health risks or, alternatively, *negligible* health risks.

Predictive risk assessment is generally accepted by the scientific community as the only currently available method for evaluating the risks posed by exposure to chemical contaminants under varying conditions. This approach and its limitations are described in detail in the literature (e. g., N.A.S, 1983a; Environ Corp., 1983). Importantly, what are deemed to be “safe” or “acceptable” levels of risk for the protection of public health involves subjective judgments, often including consideration of the costs of achieving those levels.

Predictive risk assessment has historically been applied to contaminants found in environmental media other than groundwater. Its application to groundwater is believed appropriate because many of the scientific and technical issues that motivated the use of predictive risk assessment in the past are independent of the environmental medium in which the contaminants occur (Environ Corp., 1983). Some of these issues concern the risks associated with chemical exposures that do not produce immediately observable effects or for which the nature and duration of the exposure cannot be readily

identified. At the same time, the occurrence of contaminants in groundwater raises questions that have not yet been fully examined in the context of predictive risk assessment and public health protection; these questions are related, for example, to multiple pathways of exposure.

Conducting a risk assessment for groundwater contaminants consists of four basic steps (NAS, 1983a):

1. hazard evaluation, identification of the contaminants and their toxicological characteristics;
2. dose-response assessment, i.e., specification of the “no observed effect level (NOEL) for non-carcinogens and of the unit risk for carcinogens;
3. exposure assessment, i.e., identification of the pathways of exposure, dosage, concentration levels, and exposed population; and
4. risk characterization, i.e., translation of the above three steps into a determination of health risks.

Each of these steps is described and analyzed below in the context of groundwater. Ultimate determination of risks requires that *each* of the four steps be carried out

Hazard Evaluation

Hazard evaluation involves collecting and assessing information about the inherent toxic properties of contaminants. There are two principal sources of informa-

tion about toxic properties: 1) epidemiological or clinical studies and 2) experimental data. Molecular structure is presently of only limited value in predicting the toxic properties of chemicals (Environ Corp., 1983).

The limitations of epidemiological investigations in providing information about the toxic properties of chemicals are well described elsewhere (Environ Corp., 1983). In the context of groundwater contamination, the limitations would include:

- *Difficulties in providing proper controls on studies so that strict cause-effect relationships can be established:* Because there is so little experience in conducting epidemiological studies in the context of groundwater, there are many unresolved methodological issues concerning controls including removing sources of bias (e. g., effects of diet, cigarette smoking, and occupation), accounting for exposure to mixtures of contaminants that are also site-specific and time-varying, identifying suitable control groups, and detecting small but potentially important risks when small numbers of people are involved.
- *Difficulties in obtaining accurate data on the nature, intensity, and duration of exposure, especially when multiple chemicals are present at low concentrations:* Many contaminants are present in groundwater at low concentrations (e. g., parts per billion), and exposure may occur over long periods.
- *Difficulties in linking adverse health impacts that are observable only after long latency periods to exposure:* There is a general lack of data concerning possible health impacts on humans exposed to groundwater contamination. One systematic health investigation that was specifically oriented to groundwater suggested a relationship between high levels of carbon tetrachloride and liver damage in Hardeman County, TN (Clarke, et al., 1982, cited in Harris, 1983); however, this study involved a relatively short latency period and was not a controlled epidemiological study. Epidemiological studies related to drinking water include a set of studies that are inconclusive about an association between cardiovascular disease and chlorinated drinking water (see NAS, 1980) and studies suggesting an association between chlorinated drinking water and certain cancers (Crump, et al., 1980, cited in Harris, 1983). A recent study linked rates of leukemia and birth defects with the presence of chloroform and TCE in two wells in Woburn, MA (*Science News*, 1984).
- *Difficulties in applying the epidemiological methodology to newly introduced chemicals:* Although relatively few chemicals are widely used commercially, approximately 1,000 new chemicals are introduced into commercial production each year.

- *Difficulties in interpreting self-reported symptoms:* Self-reporting of symptoms is one of the earliest clues to a possible relationship between exposure and health impacts and can provide the basis for the design of testable, controlled epidemiological investigations. Evidence for a relationship is strong if reported symptoms are highly specific and unusual and appear to occur in "clusters. Even so, such evidence does not constitute proof of a causal link between exposure and reported symptoms. At best, reported symptoms can be checked for consistency with known hazards and serve to strengthen or weaken inferences about suspected relationships. If reported symptoms are vague and/or common (e.g., headaches, nausea, and rashes), it is unlikely that epidemiological studies will be of value (Environ Corp., 1983).

Because of the types of problems associated with epidemiological investigations, 'it is likely that most epidemiological investigations of populations exposed to groundwater contaminants would lead to inconclusive results, and there appears to be little prospect for improving this situation; these problems are inherent to methods of epidemiology' (Environ Corp., 1983). However, when populations have large exposures to high concentrations of organic chemicals, such as in Hardeman County, epidemiological investigations may be able to document adverse health impacts. In addition, when epidemiological data are supplemented with laboratory data, the likelihood of establishing cause-effect relationships can increase (Harris, 1984).

In addition to epidemiological studies, a second major source of information about toxicity is experimental data. Toxicity data derived from laboratory experiments on animals have several advantages over epidemiological and clinical investigations: exposures can be controlled, biological changes can be examined in detail, and causal relationships between exposure and toxicity can be established with high certainty.

The applicability of animal data to humans depends on the assumption that biological activity is similar among various mammalian species. There appears to be substantial evidence to support the inference of human health effects based on results from animal studies (Environ Corp., 1983); and consequently, animal data have historically been the principal sources of toxicity data for assessing the risks of chemicals (e. g., pesticides, food and color additives, and drugs) prior to their commercial introduction. Nevertheless, inferences about human health effects from animal data are still controversial. In addition, although efforts are underway to develop toxicity data for various purposes (e.g., toxicity data are available from the National Toxicology Program of the Department of Health and Human Services), OTA'S analysis suggests that a com-

plete, uniform data base for all potential groundwater contaminants is unlikely for many years (Environ Corp., 1983).

Dose-Response Relationships

The second step in a predictive risk assessment is describing dose-response relationships. These relationships link known exposure characteristics with the frequency at which toxic effects appear in exposed populations. In general, for a given duration of exposure, the frequency at which toxic effects appear in an exposed population increases with increasing dosage; in many cases, the toxic effects will become more severe as exposure increases (Environ Corp., 1983).

There are various ways to express dosage. The most common is weight of the contaminant taken into the body per unit of body weight of the exposed recipient per unit of time (e.g., milligrams (mg) per kilogram (kg) per day). Because epidemiological studies rarely provide the exposure data necessary for determining exposure characteristics, experimental data are the primary source of dose-response information.

In practice, inferences must often be made about the dose-response function for groundwater contaminants because doses are often below the range at which experimental dose-response relationships can be observed. Some cases of contamination, however, do involve exposures in the range for which experimental dose-response relationships have been determined (Harris, 1984). When the relationships can be determined, the dose-response for non-carcinogens is described in terms of the threshold dose at which no adverse response is observed, the "no observed effect level" (NOEL). For carcinogens, which do not appear to act according to a threshold concept, experimental data are used to establish a relationship between dose and carcinogenic risk known as the "unit risk, e.g., the fraction of a group of experimental animals exposed to carcinogens that develop tumors during the experiment minus the fraction of animals in the untreated (control) group that develop the same types of tumors. In general, experimentally derived measures of dose-response should be interpreted with care in estimating human dose-response relationships (Environ Corp., 1983).

¹ For example, human thresholds are probably lower than experimentally derived NOELS both because the human population is genetically more diverse and thus likely to have a broader range of susceptibilities than laboratory animals, and because the human population is exposed to a broad range of additional environmental agents. Further, because only relatively small numbers of animals can be used in carcinogenicity experiments, the experiments often involve high doses of agents, extrapolating the results to human exposures from environmental carcinogens thus involves prediction of low dose risk from high dose/high risk data.

Exposure Assessment

Exposure assessment involves determining the magnitude and duration of exposure to environmental agents. It requires estimating the dosage of contaminants received by exposed populations, identifying the exposed population, and identifying the body sites at which toxic effects are produced.

The dosage of contaminants received by exposed human populations can be estimated if information is available about both concentration levels and the intake (e.g., duration, frequency, and amount) of contaminants at given concentration levels. Determining the intake of groundwater contaminants, however, is difficult because of the multiplicity of pathways along which the contaminants can expose populations (see ch. 2).

In practice, information is most often not available about the dosage received along these different pathways, and health scientists often assume standard average values when carrying out exposure assessments. Only for the direct ingestion of contaminants via drinking water are there standard approaches for estimating dosage. Although there appears to have been little attempt thus far to conduct comprehensive exposure analysis (Environ Corp., 1983), approaches for incorporating the different possible pathways of exposure have been discussed within the scientific community.²

Table A. 1.1 lists the types of data and assumptions that would be necessary to estimate dosage from each possible route of exposure to groundwater contaminants. Because many of the parameters shown in table A. 1.1 vary from site to site and thus cannot be readily standardized, exposure assessments will probably have to be made at the site-specific level. Further, daily concentrations of organic chemicals in groundwater can fluctuate by more than an order of magnitude. Accurate average exposures can be calculated only if a monitoring program is designed to account for this fluctuation; most monitoring data currently available are not adequate for calculation of accurate average exposure (Harris, 1984). This difficulty argues for careful site analysis of contaminant concentrations, soils, and the habits of the exposed populations.

Identification of exposed populations is important because different people exhibit different susceptibilities to a toxic agent. In most cases, the general population would be exposed and would exhibit the full range of susceptibilities. At some sites, however, principally

²For example, in the risk assessments conducted by the Safe Drinking Water Committee of the National Research Council (NRC), safe drinking water exposure limits were estimated on the basis of an arbitrary assumption that only 20 percent of a person daily exposure to a contaminant would come from the direct ingestion of water. (See also NAS, 1983a, NRC, 1980)

Table A-1.1.—Data and Assumptions Necessary To Estimate Human Dose of a Groundwater Contaminant From Knowledge of its Concentration in Groundwater^a

1. Direct ingestion through drinking:
 - Amount of water consumed each day (generally assumed to be 2 liters for adults and 1 liter for a 10 kg child).
 - Fraction of contaminant absorbed through wall of gastrointestinal tract.
 - Contaminant concentrations.
 - Average human body weight.
2. Inhalation of contaminants:
 - Air concentrations resulting from showering, bathing, and other uses of water.
 - Variation in air concentrations over time.
 - Amount of contaminated air breathed during those activities that may lead to volatilization.
 - Fraction of inhaled contaminant absorbed through lungs.
 - Average human body weight.
3. Skin absorption from water:
 - Period of time spent washing and bathing.
 - Fraction of contaminant absorbed through skin during washing and bathing.
 - Average human body weight.
4. Skin absorption from contaminated soil:
 - Concentrations of contaminant in soil that has been exposed to contaminated groundwater.
 - Amount of daily skin contact with soil.
 - Amount of soil ingested per day (e.g., by children).
 - Absorption rates (e.g., by skin and gastrointestinal tract).
 - Average human body weight.
5. Ingestion of contaminated food:
 - Concentrations of contaminant in edible portions of various plants and animals that have been exposed to contaminated groundwater.
 - Amount of contaminated food ingested each day.
 - Fraction of contaminant absorbed through wall of gastrointestinal tract.
 - Average human body weight.

^aThe total dose is equal to the sum of the doses from the five routes.

SOURCE: Environ Corp., 1983.

subgroups will be exposed (e. g., children and the elderly), and they may exhibit specific susceptibilities.

Another aspect of exposure assessment involves identifying the body site at which toxic effects are produced. For example, some contaminants produce their toxic effects directly at the point of contact (e.g., the skin, lung, and gastrointestinal tract). If contaminants are to produce effects at internal body sites (systemic effects), they must first pass through physical barriers—i. e., the gastrointestinal wall, the skin, or the lungs. The rate and amount of absorption vary from contaminant to contaminant; these data are most frequently not available. In the absence of data from human subjects, the common practice among public health scientists is either to adopt absorption rate values from experimental stud-

ies of substances having similar chemical and physical characteristics or to assume that absorption is complete along every pathway (Environ Corp. , 1983).

Risk Characterization

The fourth and last step in the risk assessment process is risk characterization. Once information is obtained about contaminant toxicity, dose-response relationships, and exposure, the risk faced by exposed populations can be determined.

With respect to non-carcinogens, common practice is to:

1. calculate an acceptable daily intake (ADI) level by dividing the experimentally determined NOEL by a safety factor (to account for uncertainties in the measurements);
2. modify the ADI if exposure routes other than ingestion are to be considered; otherwise incorporate additional safety factors; and
3. calculate the margin-of-safety (MOS) by dividing the experimental-NOEL by the actual dose and compare the MOS to the safety factors used in calculating the ADI. (Note that the lower the value of the MOS, the larger the risk to the exposed population.)

For carcinogens, risk is characterized by multiplying the actual daily lifetime dose by the unit risk. Although an explicit estimate of risk is obtained, this estimate still embodies uncertainty and is treated (e. g., by FDA and EPA) as an upper limit of the true risk.

The ADI and the MOS for non-carcinogens and the acceptable risk for carcinogens are designed to ensure that exposed populations are not at significant risk. Although the calculation of these values for any given contaminant involves many simplifying assumptions and approximations, an additional limitation is that these estimates treat contaminants individually and independently of each other. In most instances, however, populations are exposed not to individual contaminants but to complex and possibly time-varying mixtures.

How and where contaminants interact with each other to produce toxic effects are complicated and poorly understood; some evidence suggests that such interactions are significant.³ The health risks from exposure to combinations of contaminants may differ either qualitatively or quantitatively from health risks from exposure to individual contaminants. Although such interactions are

³Examples include the marked synergism between cigarette smoking and asbestos in the induction of lung cancer, the reaction of secondary amines and nitrites in the stomach to form carcinogenic nitrosamines, and the synergistic effects between alcohol and halogenated hydrocarbons (e. g., carbon tetrachloride) to cause liver damage (see Environ Corp., 1983, for complete references).

not unique to groundwater, they do pose a significant impediment to reaching conclusions about acceptable levels of exposure to groundwater contaminants (Environ Corp., 1983).

There are no generally applicable protocols for testing the effects of contaminant interactions, and there are few data to guide the development of such protocols. For now, risk assessments that are to take into account possible interactions must be based on considerations

other than empirical evidence. Although the potential importance of interactions is recognized, especially with respect to groundwater, there is no area of standard setting that has taken interactions into account as a matter of course.⁴

⁴EPA has considered treating carcinogenic risks as additive, i.e., that the total carcinogenic risk is equal to the sum of the risk of each of the individual contaminants (Environ Corp., 1983).

[illegible]

CONTAMINANT	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
A.1. AROMATIC HYDROCARBONS (Continued)																								
20. Methylthioazothiazole						X	X									X					X		X	
21. 4-Methylene-bis-2-chlorobenzoic acid (MOCA)	X	X		X		X										X								
22. Naphthalene	X	X		X							X						X							
23. o-Nitroaniline									X		X					X				X				
24. Nitrobenzene						X	X									X								
25. 4-Nitrobenzoic acid					X											X								
26. n-Nitrosodiphenylamine						X																		X
27. Phenanthrene									X															X
28. n-Propylbenzene										X														
29. Pyrene		X			X	X					X													
30. Styrene (vinyl benzene)	X	X		X	X	X	X		X					X										
*31. Toluene	X	X		X																				
32. 1,2,4-Trimethylbenzene		%			X				X															
33. Xylenes (m,o,p)	X			X		X	X		X															

CONTAMINANT	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	2	22	23	24
A.2. OXYGENATED HYDROCARBONS																								
1. Acetic acid	X	X		X	X																			
2. Acetone	X			X					X															
3. Benzophenone			*																					
4. Butyl acetate																								
5. N-butyl-benzylphthalate																								
6. Di-n-butyl phthalate	X			X					X															
7. Diethyl ether	X			X					X															
8. Di-n-butyl phthalate				X																				
9. Diisopropyl ether																								
10. 2,4-Dimethyl-3-hexanol																								
11. 2,4-Dimethyl phenol																								
12. 2,4-Dimethyl phthalate																								
13. 1,4-Dioxane	X			X		X	X																	X
14. Ethyl acrylate																								
15. Formic acid	X			X					X															
16. Methanol	X					X			X															
17. Methyl alcohol																								
18. Methylcyclohexanone																								
19. Methyl ethyl ketone																								
20. Methylphenyl acetamide																								

%

[illegible]

[illegible]

[illegible]

[illegible]

[illegible]

CONTAMINANT	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	6	7	18	19	20	21	22	23	24	
B.2. NONMETALS AND ANIONS																									
*1. Ammonia	x	x		x	x	x	x		x																
2. Boron																									
*3. Chlorides																									
*4. Cyanides	x	x	x	x					x																
*5. Fluorides									x																
*6. Nitrates																									
7. Nitrites																									
8. Phosphates																									
*9. Sulfates																									
- Sulfites																									
TOTAL NUMBER OF CHEMICALS	5	5	6	13	38	37	62	5	2	67	6	17	4	16	8	6	5	3	4	7	21	20	14	31	38

Footnotes

^a Compiled from a partial survey of literature conducted by Environ Corp., 1983.

"*" indicates that the chemical is known to produce one or more of the adverse effects listed in the table (see chapter 2, the section on Adverse Impacts, and app. A.4).

Numerical key of toxic effects:

- | | |
|---|------------------------------|
| 1. Eye irritation | 13. Cardiovascular effects |
| 2. Skin irritation | 14. Gastrointestinal effects |
| 3. Allergic sensitization | 15. Cholinesterase inhibitor |
| 4. Upper respiratory tract irritation | 16. Methemoglobinemia |
| 5. Lung/respiratory effects | 17. Skin diseases |
| 6. Liver damage | 18. Visual damage |
| 7. Kidney damage | 19. Endocrine effects |
| 8. Pancreatic damage | 20. Reproductive effects |
| 9. Central nervous system (CNS) effects | 21. Embryotoxicity |
| 10. Peripheral nervous system effects | 22. Teratogenicity |
| 11. Blood cell disorders | 23. Mutagenicity |
| 12. Immunological effects | 24. Carcinogenicity |

Source: Office of Technology Assessment.

A.3 FREQUENCY OF DETECTION OF SELECTED CHEMICALS IN GROUNDWATER

CHEMICAL	SAMPLING SCHEME		
	Random	Non-random	Not specified
A.1. AROMATIC HYDROCARBONS			
Benzene		1.7-15	8.5
Ethylbenzene		0-6-44	
Fluoranthene			6e9
Propylbenzene		0.2	
Toluene		1 .0-5.2	<5.0
Xylenes		1.7-2,1	<5.0
A.2. OXYGENATED HYDROCARBONS			
Acetone		206	
Butyl acetate			< 5.0
Di-n-butyl phthalate			28-6
Dichlorophenol			17.2
Diethyl phthalate			14.3
Methyl ethyl ketone			(5.0
Phthalic acid			21.4
A.3. HYDROCARBONS WITH SPECIFIC ELEMENTS			
Bromobenzene		0.4	
Bromodichloromethane		50.9	69.2
Bromoform		30.9	36.3
Carbon tetrachloride		3.1-7.4	5-50
Chlorobenzene		0.2	7.1
Chloroform		11-53.2	70.3
Chloromethane (Methyl chloride)		3.7	

CHEMICAL	SAMPLING SCHEME		
	Random	Non-random	Not specified
Chloroto luene		0.2	
Dibromochlorocxnethane		46.3	64.5
Dibromochloropropane (DBCP)		2.6	
Dichlorobenzene		0.8	12.9
Dichloriodomethane		2.7	30.3
1,1-Dichloroethane		1.9-23.1	1-34
1,2-Dichloroethane	1.1-7.0	1.5-17.1	2-73
1, 1-Dichloroethylene		3.1	7.1
1,2-Dichloroethylene		4.8-38.5	7.1-21.4
Dichloromethane			607
2,4-Dichlorophenol			17.2
1,2-Dichloropropane		1.5	
Ethyl chloride			7.1
Malathion			7.1
Methyl parathion			7.1
Pentachlorophenol (PCP)			6.9
Polychlorinated biphenyls (PCB)		7.8	
Tetrachloroethylene		2.1-9.4	2-34
Trichloroethanes (TCA)	4.3-8.1	8.1-15.8	2-66
T'r ichloroethylene (TCE)	1.7-11.3	3.6-50.1	2-79
Vinyl chloride		1.3	1-36

Source: Office of Technology Assessment; University of Oklahoma, 1983.

A.4 SUBSTANCES IN GROUNDWATER WHOSE DETECTED CONCENTRATION HAS EXCEEDED STANDARDS AND TYPES OF STANDARDS EXCEEDED

SUBSTANCE	State DW	State GW	National DW		Health Advisory			Ambient Water Quality Control
			Primary	Secondary	1-Day	10-day	Long-term	
A.1. AROMATIC HYDROCARBONS								
Benzene	X	X				X	X	X
Ethyl benzene								X
Toluene	X	X				X	X	
A.2. OXYGENATED HYDROCARBONS								
1,4-Dioxane	X	X				X		
Phenols	X	X						
A.3. HYDROCARBONS WITH SPECIFIC ELEMENTS								
Alachlor		X						
Aldicarb	X	X						
Bromacil		X						
Bromodichloromethane								
Carbofuran	X				X	X	X	
Carbon tetrachloride	X	X			X	X		
Chloroform		X						
Dibromochloropropane (DBCP)	X	X						
Dibromoethane	X							
Dichlorobenzene (-p)								
Dichlorodiphenyltrichloroethane (DDT)		X						
1,2-Dichloroethane	X	X						
1,1-Dichloroethylene	X	X			X	X	X	
1,2-Dichloroethylene	X	X			X	X		
Dichloromethane (methylene chloride)	X	X				X	X	
2,4-Dichlorophenoxyacetic acid (2,4-D)	X	X						
Dichloropropane	X							
Dioxins		X						
Endosulfan		X						
@ -Hexachlorocyclohexane	X							
&-Hexachlorocyclohexane	X							
]liexachlorocyclohexane (F-BHC, or Lindane)		X						
Methyl parathion		X						

SUBSTANCE	State DW	State GW	National DW		Health Advisory			Ambient Water Quality Control
			Primary	Secondary	1-Day	10-day	Long-term	
A.3. HYDROCARBONS WITH SPECIFIC ELEMENTS (cent d)								
Polychlorinated biphenyls (PCBS)	X	X				X		X
RDX (Cyclonite)								X
Tetrachlorobenzene								X
Tetrachloroethane								X
Tetrachloroethylene	X	X			X	X	X	X
Toxaphene		X	X					X
1,1,1-Trichloroethane	X	X					X	X
Trichloroethylene (TCE)	X	X			X	X	X	x
Trinitrotoluene (TNT)								X
Vinyl chloride	X	X						X
A.4. OTHER HYDROCARBONS								
Gasoline	X	X						
B.1. METALS AND CATIONS								
Aluminum		X						
Arsenic	X	X	X					X
Barium		X	X					
Beryllium								X
Cadmium		X	X					X
Chromium		X	X					X
Cobalt		X						
Copper		X						X
Iron	X	x		x				
Lead		X	X					X
Manganese	X	X		X				
Mercury		X	X					X
Molybdenum		X						
Nickel		X						X
Selenium		X	X				X	
Silver		X	X					X
Sodium	X	X						
Vanadium		X						
Zinc		X		X				X
B.2. NONMETALS AND ANIONS								
Ammonia		X						
Chlorides		X		X				

SUBSTANCE	State DW	State GW	National DW		Health Advisory			Ambient Water Quality Control
			Primary	Secondary	1-Day	10-day	Long-term	
B.2. NONMETALS AND ANIONS (cent d)								
Cyanides	X	X						X
Fluorides	X	X	X					
Nitrates		X	X					
Sulfates		X		X				
D. RADIONUCLIDES								
Radium 226	X	X	X					
Uranium 238		X					X	

Abbreviations: DW = drinking water; GW = ground water.

“X” in State DW or State GW column means that the standard set by at least one State has been exceeded.

Source: Office of Technology Assessment.

A.5 SOURCES OF GROUNDWATER CONTAMINATION

This appendix was compiled to supplement and/or substantiate information summarized in chapter 2 (see table 8). Although an extensive survey of sources was attempted, time limitations precluded collecting some data. Thus the information in this appendix is that which was readily available to OTA; it should not necessarily be regarded as exhaustive or definitive.

When available and appropriate, this appendix contains the following information for each source:

- general information regarding the definition, use, and location of the source;
- details of the assumptions and calculations used in estimating the numbers of facilities or activities of a source type;
- details of the assumptions and calculations used in estimating the amount of material flowing through or stored in all facilities or activities of a source type; and
- information regarding the potential of both *individual* facilities or activities and all facilities or activities of a source type to contaminate groundwater.

Selected references on the potential of sources to contaminate groundwater are listed at the end of the appendix.

1. Subsurface Percolation: Septic Tanks and Cesspools

Septic tank systems consist of a buried tank and drainage system designed to collect waterborne wastes, remove settleable solids from the liquid by gravity separation, and permit percolation into the soil of clarified effluent. They are best suited for small volumes and periodic flows.

The highest regional densities of use in the United States occur in the eastern third of the country and along portions of the west coast (USDA, 1981a). Septic tank systems and cesspools serve more than 100,000 housing units in four counties (Nassau and Suffolk, NY; Dade, FL; and Los Angeles, CA) and more than 50,000 housing units in 23 counties (EPA, 1977a).

Development of Estimates of Numbers and Amounts

There were an estimated 19.5 million *domestic* on-site disposal systems in the United States in the mid-1970s, of which 16.6 million were septic tanks and cesspools (EPA, 1977a); presumably the remaining 2.9 million systems were privies or chemical toilets. Little information is available regarding the number of *commercial and industrial septic* tank systems. DeWalle, et al. (1980, cited in DeWalle, et al., no date) estimated that the State of Washington has at least 500 large on-

site systems serving restaurants, hospitals, and larger industrial customers. Miller (1980) estimated that 25,000 industrial septic tanks are in operation in the United States based on the number of industrial establishments using water, but no documentation for the figure was provided.

Estimates of annual flow to an individual septic tank from an average household range from 49,275 gallons per year per household (gyh) (Miller, 1980: 45 gallons per person per day X 3 persons per household X 365 days per year) to approximately 75,000 gyh (derived from information in Pye, et al., 1983: 3.5 billion gallons per day X 365 days per year ÷ 17 million tanks). Thus a minimum estimate of the total annual flow to all domestic systems would be approximately 820 billion gallons per year (49,275 gyh X 16.6 million systems), and a maximum estimate would be approximately 1,460 billion gallons per year (75,000 gyh X 19.5 million systems).

Little direct information is available about flow rates to and leakage from industrial septic tanks. Assuming that the use of industrial septic tanks is comparable to domestic systems, there could be an estimated annual flow of approximately 1.2- 1.9 billion gallons (minimum estimate: 49,275 gallons per year X 25,000 systems; maximum estimate: 75,000 gallons per year X 25,000 systems).

The range of estimates for domestic systems is probably very near to the actual amount because the underlying assumptions and data are based on studies of domestic systems (e. g., data are cited in: EPA, 1977a; Miller, 1980; Pye, et al., 1983). The estimates for industrial systems could be incorrect by more than 100 percent because information is lacking on annual flow to individual systems and no systematic surveys of numbers have been conducted on a nationwide basis.

Potential for Groundwater Contamination

Of all the sources known to contribute to groundwater contamination, septic tank systems and cesspools directly discharge the largest volume of wastewater into the subsurface. They are also the most frequently reported source of contamination (EPA, 1977a), and they contribute to both local and regional problems. Contaminants are principally from human wastes and household piping systems and include: nitrate, chloride, and coliform bacteria (e. g., DeWalle, et al., 1980); various metals (e. g., lead, zinc, copper, manganese, tin, and iron; Miller, 1980); viruses (Hain, et al., 1979); and others (e. g., see Miller, 1980).

The estimates of total annual discharge represent the potential volume of leachate released from the source. These figures are not equal to the volume of contaminated wastewater reaching groundwater because of ren-

ovative capacities of the soil system and evaporative losses from septic tank drain fluids (which occur even though the tanks are located in the soil) (Canter, et al., 1983).

Major factors affecting the potential of septic systems to contaminate groundwater in general are the density of systems per unit area and hydrogeological conditions. Areas with a density of more than 40 systems per square mile are considered regions with potential for contamination (EPA, 1977a); based on this criterion, portions of the Eastern United States and California exhibit the greatest potential for contamination. Local problems with septic tank systems can occur when individual systems are overloaded or when additives (e. g., TCE) are used to clean and unclog septic lines. Experiments conducted in Suffolk County, NY, confirm that organic cleaning solvents can leach from cesspools into groundwater (Andreoli, et al., 1980). Approximately 400,000 gallons of septic tank cleaning fluids (containing TCE, benzene, and dichloromethane (methylene chloride)) were used by homeowners in 1979 on Long Island alone (Burmaster, et al., 1982).

The design lives of septic tank systems are typically 20-40 years, after which time deterioration is likely. Design considerations for the percolation of effluent relate to the soil absorption system: the flow regime, the storage and carrying capacity of the receiving soil, the attenuation capacity of the biological mat in the leaching field, the subsurface soil type, and depth to the water table (Laak, et al., 1974).

2. Injection Wells

Several types of injection wells are used to inject or discharge wastes into or perform other functions in the subsurface:

- hazardous waste wells;
- non-hazardous waste wells (e. g., brine injection wells, and agricultural, urban runoff, and sewage disposal wells); and
- non-waste wells (e. g., wells for enhanced oil recovery, artificial recharge, in-situ recovery, and solution mining).

Hazardous waste wells are highly localized but can be expected to be regionally concentrated near industrial generators of these wastes.

Among the *non-hazardous waste wells*, agricultural wells are located in farming areas while urban runoff and sewage disposal wells are located primarily in urban areas. Because brine is a byproduct of oil production, brine injection wells are located primarily in areas of oil and gas production (e. g., the Southwest, Louisiana, Pennsylvania; University of Oklahoma, 1983).

Among the *non-waste wells*, enhanced oil recovery (EOR, also known as tertiary) wells follow a distribution pattern similar to that of oil production wells. Artificial recharge wells are usually located in areas of limited or vulnerable groundwater supplies; two major areas are in the High Plains (Ogallala Aquifer) and in coastal areas (e. g., to minimize salt-water intrusion). In-situ recovery wells are generally located in the oil shale regions of the Rocky Mountains. Solution mining injection wells are generally associated with uranium resources in the Southwest.

Development of Estimates of Numbers and Amounts

Hazardous Waste Wells.—Injection wells used primarily for hazardous waste disposal numbered approximately 280 in 1973 (Pye, et al., 1983). In 1981, 8.6 billion gallons of hazardous wastes were disposed of at 87 injection well sites (Dietz, et al., 1984).

The total number of injection wells is not known, and the validity of extrapolating data from strictly hazardous waste injection wells to all injection wells (even if most of them are used for hazardous waste disposal) is questionable. Other data indicate that as much as 11 percent of the Nation's liquid wastes may be disposed of in underground injection wells (Feliciano, 1983).

Brine Injection Wells and Enhanced Oil Recovery Wells.—Brine injection wells and enhanced oil recovery (EOR) wells are treated together here (and separately from non-hazardous waste wells and non-waste wells, respectively) for two reasons. First, more information is available for these wells than for other non-hazardous waste and non-waste wells. Second, EOR wells are injection wells used in tertiary oil production, and brine often is the injection fluid used in the EOR process.

In the early part of the century, most brine was disposed of in simple pits and caused many groundwater problems. Most States now ban the disposal of brine in pits, so most brine is disposed of in injection wells; illegal brine dumping into pits and streams and onto roads is a problem in some areas (e. g., Ohio; Dalton, 1983).¹ In recent years, at least 17 States have reported brine-related contamination incidents (Miller, 1980). For example, in Texas in the 1960s, approximately 69 percent of brine was reinjected, 21 percent was disposed of in pits, and 10 percent was discharged onto surface

¹ Illegal brine dumping may be prevalent in some areas of the country. For example, Dalton (1983) states that excessive brine is often dumped on roads for dust control, beyond legal limits, and that some companies have been observed dumping brine directly into streams. However, the Ohio Oil & Gas Association (cited by Abbott, 1983) contends that some brine is legally used on roads for dust control and disputes the allegations of illegal dumping

water; and approximately 23,000 contamination incidents were reported (University of Oklahoma, 1983).

Miller (1980) estimated that 60,000 brine injection wells were in operation in the 1970s. A recent report indicated that 140,000 injection wells are used either for disposal of brine fluids brought to the surface during oil and gas production or for the injection of fluids in EOR processes (Kaplan, et al., 1983). EPA (1983a) listed over 119,000 EOR wells and an additional 37,000 injection and disposal wells (not all of which were used for brine disposal or EOR processes) in its Federal Underground Injection Control Reporting System (EPA, 1983a). Given these figures, it seems reasonable to conclude at this time that the number of brine disposal and EOR wells totals approximately 140,000.

Miller (1980) also estimated that approximately 460 billion gallons of brine per year were disposed of in injection wells. (Note that Miller indicated 260 bgy on p. 511 but 460 bgy p. 304; 460 bgy was the figure given by Fairchild, et al., 1980, cited in University of Oklahoma, 1983). The OTA updated estimate of the amount of brine disposal is based on estimates of brine production: although varying widely in different areas and operations, approximately 4 barrels (bbls) of brine are produced for every barrel of oil produced (Kaplan, et al., 1983), and approximately 8.55 million bbls of crude oil were produced per day in 1981 (CEQ 1982). Given these figures, approximately 525 billion gallons of brine would be produced annually (8.55 million bbls oil per day X 4 bbls brine/bbl crude oil X 365 days per year X 42 gallons/bbl), and most of the brine is injected into wells.

The current level of oil produced from EOR processes is approximately 400,000 bbls/day (Kaplan, et al., 1983). The number of barrels of water injected per barrel of oil produced varies greatly depending on the particular EOR production process (Royce, et al., 1982). Assuming that 4 bbls of water are injected per barrel of oil produced (this figure is well within the range of figures presented in Royce, et al., 1982), then approximately 24.5 billion gallons of water per year would be injected in EOR processes (400,000 bbl per day X 4 bbl water per bbl oil X 365 days per year X 42 gallons/bbl).

Non-hazardous Waste Wells (excluding brine disposal wells).—Miller (1980) stated that at least 40,000 agricultural, urban runoff, and sewage disposal wells were in operation but that this estimate was probably much too low. For example, Miller cited 15,000 such wells in Florida; information obtained for OTA'S study indicates there may be as many as 10,000 runoff wells in Phoenix, AZ (University of Oklahoma, 1983). Kaplan, et al. (1983) estimated that approximately 500,000 injection wells are in existence, of which approximately

140,000 are used in brine disposal or EOR processes; thus there would be approximately 360,000 other disposal wells in operation, presumably for agricultural, urban runoff, and sewage disposal purposes. It is not possible at this time to estimate the volumes of materials flowing through these wells. An on-going EPA inventory of Class V injection wells (e. g., surface water drainage, air-conditioning return, and other wells) will not be completed at least until 1985 (Anzzolin, 1983).

Non-waste Wells (excluding EOR wells).—At least 12,000 solution mining wells (including sulfur mining via the Freische method) are in operation (EPA, 1983a). No information was available regarding the amounts of materials involved in these operations.

Potential for Groundwater Contamination

EPA (1979) estimated that at least 21,000 injection wells in the United States require corrective action. Although injection wells can be constructed, operated, and monitored properly, contamination of groundwater can occur in a number of ways, primarily related to the construction, operation, and eventual closing of the wells (EPA, 1979):

1. faulty well construction (e. g., drilling and casing);
2. the forcing upward of pressurized fluids into nearby wells and groundwater formations (see below);
3. the forcing upward of pressurized fluids into faults or fractures in confining beds;
4. injection into or above usable aquifers (e. g., drinking water supplies);
5. the migration of fluids into hydrologically connected usable aquifers (e. g., drinking water supplies); and
6. faulty well closing.

The second item on the EPA list above maybe of major significance in regions where heavy oil and gas production and associated brine wells are located because it includes abandoned and poorly maintained production wells. These wells are a potential source of contamination because brines injected into disposal wells can move laterally through the injection zone into unplugged, uncapped, or abandoned wells and subsequently leak into groundwater formations (Burmaster, et al., 1982; Kaplan, et al., 1983; Thornhill, 1975). Kaplan, et al. (1983) estimated that there are approximately 1.2 million abandoned wells (production wells, and mineral exploration and testing wells; see also Gass, et al., 1977) near areas of underground injection wells and, further, that the location of many abandoned wells is not known.

Depending primarily on the quality of recharge water, artificial recharge systems can alter groundwater quality; such alterations may also change the aquifer biologically

(University of Oklahoma, 1983). Soils can be clogged by suspended matter in the recharge water and by the associated biological activity. Even the disposal of a simple waste such as air conditioning return water can degrade groundwater by raising the temperature and adding chemicals (e. g., heavy metals).

3. Land Application

Land application of treated wastewater and wastewater byproducts (i. e., sewage sludge) is often used in place of more costly disposal processes. Its primary goals are the biodegradation, immobilization, and/or stabilization of various chemicals, and the beneficial use of nutrients contained in the wastewater or sludge. The wastewater itself is applied primarily by spray irrigation. Sludge is applied on agricultural or forest lands, used as commercial compost, disposed of in landfills, and applied in land reclamation projects (e. g., for strip mine reclamation; Weiss, 1983). Sludge is also disposed of by incineration and by ocean dumping (EPA, 1983 b).

Most of the information available concerns municipal sludge characteristics and production. However, industrial sludge is sometimes disposed of in landfills. Industrial sludge includes effluent treatment sludge, stack scrubber residue, fly and bottom ash, slag, and numerous other manufacturing residues. In general, the production of sludge is concentrated around major industrial and population centers but land application is generally practiced in less populous areas (e. g., cropland) (University of Oklahoma, 1983).

Development of Estimates of Numbers and Amounts

The exact number and average size of sludge-spreading operations for municipalities is not known, but at least 2,463 publicly owned treatment facilities applying liquid or thickened sludge on land and 485 using spray irrigation were in operation or under construction in 1982 (EPA, 1983c).

About 6.8 million dry tons of sludge were produced by municipalities in 1982 (EPA, 1983b). Between 24 and 29 percent of the sludge generated in the United States is spread directly on crops (EPA, 1981b, 1983 b). Another 18-21 percent is distributed free or is marketed, and most of it is subsequently deposited on cropland. Thus 40-50 percent of the municipal sludge generated—3-4 million dry tons per year—is used in some kind of direct land application.

Data are lacking on the amounts of industrial sludge produced annually and the number of sites involved but most of it is thought to be disposed of in solid waste sites and lagoons (Miller, 1980). During 1981, 70 hazardous

waste land treatment facilities (excluding landfills) regulated by EPA under RCRA regulations treated approximately 0.1 billion gallons of hazardous wastes (Dietz, et al., 1984).

Potential for Groundwater Contamination

Groundwater contamination can occur when substances in sludge are leached by precipitation after the sludge is applied to the land. The substances of most concern include nitrogen, phosphorus, and heavy metals (EPA, 1983 b); heavy metals also can limit the use of sludge in agriculture because they can be absorbed into the cover crop (Gurnham, et al., 1979).

The rate and duration of sludge application are determined by soil types, the nitrogen, phosphorus, and heavy metal content of the wastes, length of the irrigation season, and the nutrient uptake characteristics of the cover crop (Knox, et al., 1980; Young, 1978). Most States consider land application of municipal sludge at an agronomic rate (i. e., annual rate at which the nitrogen and/or phosphorus available to the crop from sludge does not exceed the annual nitrogen and/or phosphorus requirements of the crop) to have little potential for contamination of groundwater (EPA, 1983 b). Reduction of application rates before planting and addition of nutrients near crop roots during the growing season ('side-dressing' also may alleviate some problems (Swanson, 1983). Heavy metals in municipal sewage are contributed by industry (e. g., electroplating and metal-finishing industries; other metal production, processing, and fabrication industries; and nominally non-metal industries), commercial establishments, domestic water supplies, and non-food household commodities (Gurnham, et al., 1979). The potential for contamination by heavy metals may be minimized if quality control procedures (e. g., industrial pretreatment and wastewater and sludge monitoring) are followed.

4. Landfills

The solid wastes deposited in landfills are generally classified as hazardous or non-hazardous. Hazardous solid wastes are specifically defined under RCRA regulations (see OTA, 1983a); various waste products are excluded from the definition: domestic sewage wastes, irrigation return flows, radioactive wastes, and some industrial wastes. Non-hazardous solid wastes as defined here encompass all solid wastes not included in the RCRA definition of hazardous wastes.

Solid waste products (e.g., from residences, small industries, and commercial activities) are generally deposited in municipal landfills; these wastes are usually, but not always, non-hazardous. Sanitary municipal land-

fills are landfills that are designed to minimize adverse environmental impacts (Miller, 1980). Industrial landfills are used for the disposal of solid wastes from large industries; the wastes are often hazardous.

The distribution of municipal landfills is assumed to follow the general distribution of population and thus should be concentrated around urban population centers. Most sanitary municipal landfills are small operations: about 80 percent of the sanitary landfills handle less than 50 tons of waste per day, and approximately 1 percent handle amounts in excess of 1,000 tons per day (Waste Age, 1981). Industrial landfills are probably concentrated near industrial facilities.

Development of Estimates of Numbers

The number of municipal solid waste land disposal sites is not easily determined. EPA's 1977 Report to Congress (1977a; see also Miller, 1980) estimated the number to be 18,500. This figure included not only sanitary municipal landfills but also some industrial landfills and open dumps; only about 5,600 were *licensed* sanitary landfills and most of the remaining sites were open dumps (Petersen, 1983). A recent survey estimated a total of 12,991 landfills in the United States (Petersen, 1983). These estimates included primarily sanitary municipal landfills but it also included non-hazardous industrial sites and 2,395 open dumps. Thus fewer than 10,000 sanitary municipal operations are known to be in operation (how many fewer than the 10,000 is not known because the number of industrial sites was not specified). In addition, the number of abandoned or closed municipal landfills and open dumps could be equal to the number of known sanitary municipal landfills (Eldridge, 1978). Thus a first approximation of the number of municipal landfills in the Nation might be 15,000-20,000 (fewer than 10,000 municipal landfills X 2, to account for both operating and abandoned or closed municipal landfills; see the discussion on Open Dumps, below). Conservatively, this estimate is probably correct within a range of 100 percent.

The exact number of industrial solid waste land disposal sites is not known, but EPA has estimated that there are 75,700 active landfill sites for industrial wastes (CEQ 1981 b). About 199 hazardous waste landfill facilities are known (Dietz, et al., 1984). In addition, a large portion of industrial solid wastes, including some that are considered hazardous, are disposed of in municipal landfills (Miller, 1980).

Development of Estimates of Amounts

Approximately 138 million tons of municipal solid wastes were handled by municipal solid waste disposal

facilities during 1978 (CEQ 1982). This figure is probably a relatively accurate estimate of the amount of solid wastes handled annually by sanitary landfill facilities because it is based on relatively *extensive nationwide* surveys.

Estimates of the amounts of non-hazardous industrial solid wastes and of hazardous wastes disposed of in landfills are not as accurate. The range of estimates for *non-hazardous industrial solid wastes* is 40-140 million wet tons per year. The minimum estimate of 40 million wet tons per year is derived as follows. Approximately 150 million tons of *total solid* wastes were generated by industry in 1980 (CEQ 1982), and approximately 45 million wet tons were hazardous (EPA, 1981 b); thus 105 million wet tons were non-hazardous industrial solid wastes (150 mty - 45 mty). Assuming that the proportion of solid wastes disposed of in landfills is the same for industry's non-hazardous solid wastes as it is for hazardous solid wastes (40 percent),² then the minimum amount disposed of is approximately 40 million wet tons per year (0.40 X 105 mty).

The maximum estimate of the amount of non-hazardous industrial solid waste disposal is approximately 140 million wet tons per year. This estimate is derived by applying the 40 percent rate to the higher EPA estimate of 342 million tons for non-hazardous industrial solid waste production in 1980 (EPA, 1981 b) (40 percent X 342 mty = 140 mty).

At least 0.81 billion gallons of hazardous wastes were disposed of in 199 landfill facilities in 1981 (Dietz, et al., 1984); this figure includes both liquid and solid wastes.

Utilities generate approximately 77 million wet tons of solid waste per year (EPA, 1981 b), most of which is fly and bottom ash from the burning of fossil fuels (approximately 73 million tons of ash are generated annually; OTA, 1983a). Assuming that 40 percent is disposed of in landfills, an estimated 30 million tons of solid wastes per year generated by utilities would be disposed of in landfills; the applicability of the 40 percent disposal rate assumption to utilities is not known.

Note that approximately 13-15 percent of municipal sludge produced is disposed of at landfills (EPA, 1981b; EPA, 1983 b), but this amount results in landfill disposal of only about 1 million tons per year (15 percent of the estimated 6.8 million tons of municipal sludge; see *Land Application*, above). This amount is included within the rounding errors in the above estimates.

²Approximately 40 percent of industry's hazardous *solid* wastes is disposed of in landfills of some type (EPA, 1981 b). The remainder is disposed of by (a) incineration, (b) biological treatment, (c) deep well injection, (d) land treatment, (e) resource recovery, or (f) incineration.

Potential for Groundwater Contamination

Considerations in the design of municipal landfills include the location, the area to be served, and plans for different stages in the filling process (e. g., use upon completion of the fill). Provisions must be made for controlling traffic, unloading and handling different types of wastes, placement of cover materials, fire control, control of salvage and scavenging, and monitoring. Industrial landfills have similar design, operation, and maintenance needs, although the nature of the wastes disposed of may entail additional safety considerations (hazardous waste landfills are included in this category).

Groundwater contamination can be minimized by proper design, construction, and operation and maintenance of a facility (Brunner, et al., 1972). However, facilities are not always maintained properly and some landfills are allowed to deteriorate (University of Oklahoma, 1983). Further, not all contamination controls used in landfills are effective; for example, required liners—of both natural and synthetic materials—have cracked or deteriorated when exposed to certain chemicals (OTA, 1983a). Abandoned landfills (the locations of which are not usually known to regulatory authorities) often pose a threat to groundwater quality because geologic and hydrologic characteristics were not considered in the original site selection; the same may be true for some active landfills. Many abandoned landfills were located in sand and gravel quarry pits or in environmentally sensitive areas such as marsh lands. Only 1,609 of almost 13,000 landfills surveyed reported having monitoring systems for groundwater, leachate, and/or gas in 1983 (Petersen, 1983).

Leachate generation varies with time over a facility's life, so the age of facilities could affect the amount and strength of the leachate. In addition, the amount of leachate leaving the more recent facilities could be significantly less than at older facilities. Many older landfills were not lined; and leachate collection and treatment have become common practices at a number of the more recent facilities (in the last 10 years).

Unless moisture can be totally prevented from entering a landfill, leachate will eventually be generated. Once a landfill system reaches its disposal capacity, leachate generation is directly related to the volume of water added to the system (University of Oklahoma, 1983). Leachate generation also depends on the initial moisture content of the wastes, the landfill density, the rate of filling, and infiltration water quantities. Infiltration from the surface is not the only source of water coming into a landfill; although undesirable, some landfills intersect aquifers, thereby creating another source of moisture for leachate generation.

Techniques for estimating the amount of leachate generation from landfills vary widely in their results.

Assumptions that affect the estimates include the choice of runoff coefficients, the moisture storage capacity of the waste, and evapotranspiration rates. Lu, et al. (1981) found that the error range of 25 different methods for predicting leachate generation was 1.3-5,400 percent (as reported in University of Oklahoma, 1983).

Even if the amount of leachate generated is known, not all of it reaches the groundwater. Depending on soil type and the position of the water table, the soil underlying the wastes will be able to attenuate or renovate some leachate before it reaches the groundwater. In order to develop accurate estimates of the potential for leachate to contribute to groundwater contamination, estimates must include a percentage reduction for absorption and attenuation.

5. Open Dumps

A dump is a land disposal site where solid wastes are deposited indiscriminately, with little or no regard for the design, operation, maintenance, or esthetics of the site. In an "open" dump, the wastes are almost always left uncovered. Most often the open dump is not authorized and there is no supervision of dumping (Brunner, et al., 1971, cited in University of Oklahoma, 1983). Virtually every type of solid waste has been deposited in open dumps—abandoned tires and automobiles, old furniture and kitchen appliances, industrial and commercial wastes, agricultural byproducts, trees, vegetation, demolition and construction wastes, and various household wastes—and virtually every type of topography has been used for this dumping. Open dumps are frequently burning dumps as well, whether resulting from deposition of smoldering wastes, spontaneous ignition, or intentional ignition to reduce volume.

EPA listed approximately 1,950 open dumps in its inventory (EPA, 1982a); in a more recent survey by *Waste Age* (Petersen, 1983) the figure is 2,396. Because these two estimates include only the open dumps known to regulatory authorities, they are minimum estimates. It is not possible at this time to generate any reasonable estimate of the amount of material disposed of in open dumps annually.

6. Residential (Local) Disposal

A variety of hazardous and toxic substances are commonly found in household wastes. These wastes often are disposed of in specific facilities designed for waste disposal or discharge (e. g., municipal landfills). However, they also are disposed of indiscriminately, without supervision, in gutters, sewers, storm drains, and backyard burning pits—these practices constitute resi-

dential (or local) disposal. The pattern of residential disposal follows population density and distribution.

Household wastes are composed of a wide range of product materials: pesticides; paint products (e. g., oil-based paints, thinners, removers, and wood preservatives); cleaners (e. g., drain cleaners, furniture polish, air fresheners, floor wax, disinfectants, chlorine bleaches, degreasers, nail polish removers, spot removers, oven cleaners, drycleaning fluids, detergents, aerosol sprays, rug cleaners, and shoe care products); automobile products (e. g., antifreeze, waste oil, and brake fluid); asphalt and roofing tar; and batteries.

Development of Estimates of Numbers and Amounts

Little quantitative information is available about where most household substances are ultimately disposed of, primarily because household wastes do not usually come under Federal and State regulations and are not investigated systematically. A few community and government agencies have attempted to tackle this problem; among the most noteworthy are efforts of the Water Quality Division of Seattle (Ridgley, et al., 1982), the Metropolitan Area Planning Council of Boston (MAPC, 1982), and community grassroots collection campaigns like the ones in Lexington, MA (Watson, 1983) and Seattle (Ridgley, et al., 1982).

Some quantitative information is available. Approximately 30,000 tons of household cleaners were used by the 1.2 million people in King County (Seattle Metropolitan Area) in 1980 (Ridgley, et al., 1982). The city of Tacoma, WA (population 150,000), uses 264 tons of liquid household cleaners, 72 tons of toilet bowl cleaners, and 66 tons of motor oil per year (based on Tacoma-Pierce County Health Department, no date). If the rates of use of household cleaners are extrapolated to the entire United States, then approximately 0.4-5.6 million tons of such cleaners are used annually.

Over 90 percent of households in the United States use pesticides in the home, garden, and/or yard (Savage, et al., 1980, cited in Ridgley, et al., 1982). It is estimated that 5-10 percent of all pesticides used are applied in this manner (Seiber, 1981; EPA, 1980a). The lower percentage (i. e., 5 percent) is derived as follows: at least 80 million pounds of pesticides were used in homes and gardens in 1980 (EPA, 1980b), and this figure is about 5 percent of the 1.5 billion pounds of pesticides produced annually (see *Pesticide Applications* below). The mean rate of pesticide applications by households has been estimated to be 5.3-10.6 pounds per acre, and urban soils often have higher levels of pesticide residues than do croplands (vom Runkler, et al., cited in Grier, 1981-82).

Potential for Groundwater Contamination

Residential disposal has great potential for contaminating groundwater. Uncontrolled burning can cause toxic fumes, and the hazardous materials concentrated in ashes can be leached into groundwater. Spilled oil, pesticides, and fertilizers are washed off driveways, yards, and gardens into storm drains and local streams. Toxic wastes are often poured down household drains; the result is corroded pipes (which can cause higher heavy-metal concentrations in sewage), septic tank malfunctions, pipeline leakage (including from sewers), and interference with the operation of municipal sewage treatment facilities. All these negative impacts can lead to groundwater contamination. In addition, household hazardous wastes that are deposited in specific facilities designed for waste disposal (e. g., landfills) have the potential to contaminate groundwater.

7. Surface Impoundments

Surface impoundments are used by both industries and municipalities for the retention, treatment, and/or disposal of both hazardous and non-hazardous liquid wastes. They can be either natural depressions or artificial holding areas (e. g., excavations or dikes); the term "pond" is commonly applied to a small impoundment used by industries, municipalities, agricultural operations, or households for special purposes (e. g., farm waste storage, industrial wastewater storage, and sludge disposal). The wastewater in impoundments is treated by chemical coagulation and precipitation, pH adjustment, biological oxidation, separation of suspended solids from liquids, and reduction in water temperature. Surface impoundments operate under one of two schemes: discharging and non-discharging. Discharging impoundments are designed to release their liquid contents either periodically or continuously into streams, lakes, bays, or the ocean. Non-discharging impoundments lose their liquid by evaporation and/or seepage. Impoundments that rely on evaporation are usually lined with low-permeability materials to prevent seepage and are most effective in arid areas.

Surface impoundments vary in shape, and they are operated individually or as a series (EPA, 1982 b). They range in depth from 2-3 feet (0.6-0.9 m) to more than 30 feet (9 m) below the land surface, and their surface area varies from a few tenths of an acre to thousands of acres. Agricultural, municipal, industrial, and oil and gas production impoundments are generally small—90 percent or more are under 5 acres (EPA, 1982 b). The largest impoundments reported to EPA for the agricultural, municipal, and oil and gas production categories

were 665, 850, and 79 acres, respectively. Industrial impoundments, in contrast, can be quite large—20 impoundments larger than 1,000 acres were reported to EPA, with one covering 5,300 acres. The size of mining impoundments depends on the type of mining. Ninety percent of coal mine impoundments are less than 5 acres; the largest is 293 acres. However, the surface impoundments of only 58 percent of metal mines and 48 percent of other non-metal mines are less than 5 acres; the largest in these categories are 1,990 and 1,229 acres, respectively.

Surface impoundments are located in proximity to the activity creating the liquid wastes. Thus agricultural impoundments tend to be concentrated in the Central, Midwestern, and Southeastern United States. Municipal impoundments are associated with population centers and are most common in the East. Industrial impoundments are most common in the East and Northeast, and along the Great Lakes and the west coast. Oil and gas impoundments are concentrated in Texas, Oklahoma, and Louisiana. Mining impoundments are concentrated in coal mining areas (e. g., Pennsylvania, Ohio, and West Virginia).

Development of Estimates of Numbers

As part of implementing the Safe Drinking Water Act (1442(a)(8)(C)), EPA initiated a nationwide Surface Impoundment Assessment in 1978 (EPA, 1978, 1982 b). Most of the available information about surface impoundments is the result of these efforts. Unless otherwise stated, the discussion that follows is based on the report issued in 1982.

A total of 180,973 impoundments was located by EPA: 27,912 industrial, 37,185 municipal, 19,437 agricultural, 25,038 mining, 65,488 oil and gas brine pit, and 5,913 other impoundments. The most important industrial users of impoundments are the food processing and chemical industries, each with more than 4,000 known impoundments. Other heavy industrial uses (i.e., using more than 1,000 impoundments) are for petroleum refineries; power plants; paper products; stone, clay, and glass products; primary metals; and fabricated metals. Municipal impoundments are located at landfills and water and waste treatment facilities; about 33,000 were at sewage treatment plants. Agricultural impoundments are used in crop production, animal husbandry, and other farming operations; most of them are associated with feedlot waste operations. Mining impoundments are associated with ore extraction and treatment, washing, and sorting processes. All of the numbers cited are thought by EPA to be conservative, especially for industry and for oil and gas brine pits—the estimate for oil and gas impoundments does not include burn pits, cuttings pits, or mud pits. Further, at least 1,078

impoundments regulated under RCRA were used for the storage, treatment, or disposal of hazardous wastes in 1981 (Dietz, et al., 1984). Whether these facilities are included in the total of 180,973 is not known.

Development of Estimates of Amounts

The amount of liquid wastes disposed of in surface impoundments can be estimated in a variety of ways. Approximately 50 billion gallons of liquid wastes per day are deposited in industrial surface impoundments in the United States (EPA, 1980, cited in U.S. House of Representatives, 1980), and approximately 82 billion gallons per day are deposited in all types of impoundments (The Conservation Foundation, 1982). The amount of wastes actually contributing to groundwater contamination depends on leakage from the impoundments; the commonly used leakage rate of 6 percent (Miller, 1980) is used here. Accordingly, approximately 1,095 billion gallons per year (bgy) and 1,800 bgy of liquid waste leachate from industrial and from all types of surface impoundments, respectively, are available for entry into groundwater (i. e., 50 billion gallons per day X 365 days per year X 0.06 for industry; 82 billion gallons per day X 365 days per year X 0.06 for all types),

The amount of liquid wastes deposited in municipal impoundments can also be estimated. EPA (1978) calculated that 6,300 municipal impoundments had a total flow of 4.2 billion gallons per day. Using these figures to obtain a flow rate per impoundment and applying the 6 percent leakage rate yields an estimate of 540 bgy for the 37,185 municipal impoundments found by EPA. A second estimate, of 705 bgy for municipal impoundments, can be derived by subtracting the 1,095 industrial bgy from the 1,800 total bgy; this figure is a maximum estimate because it includes all but industrial impoundments.

Brine pits are almost universally banned in the United States, but they were the major means of brine disposal prior to the 1970s. Current disposal rates for brine pits cannot be estimated because they are not monitored.

The metals mining industry puts approximately 250 million tons of tailings into ponds each year.

Thus estimates can be developed for the amount of liquid wastes converted into potential leachate for industrial, municipal, and mining impoundments and for all impoundments together. The latter figure, 1,800 bgy, is in marked contrast with Miller's (1980) estimate of 161 bgy. Miller's estimate for liquid wastes consists of separate estimates of 100 bgy from industrial treatment lagoons, 43 bgy from brine pits and basins, and 18 bgy from municipal treatment lagoons. Miller's estimate is almost certainly much too low, but the accuracy of the 1,800 bgy estimate is difficult to evaluate.

The above estimates refer to hazardous and non-hazardous liquid wastes in all surface impoundments. Quantitative information is also available regarding the deposition of hazardous liquid wastes (which may include non-hazardous liquid wastes) into surface impoundments regulated under RCRA (Dietz, et al., 1984). In 1981, 5.1 billion gallons of hazardous wastes were disposed of, 16.6 billion gallons were treated, and 14.1 billion gallons were stored in these surface impoundments (Dietz, et al., 1984).

Potential for Groundwater Contamination

In terms of their numbers and the amounts of wastes associated with them, waste impoundments (including pits, ponds, and lagoons) may be one of the **biggest** threats to groundwater. More than 23,000 cases of groundwater contamination have been documented in Texas alone, primarily resulting from brine pits (EPA, 1977a). In Colorado, 37 percent of the known impoundments pose an “actual threat” to groundwater and over 53 percent pose a “potential threat” (*The Groundwater Newsletter*, 1983a). The potential for health effects is highly variable and depends on public use of affected aquifers; most mining, oil and gas, and agricultural sites are located in remote areas and thus are likely to have a low potential for affecting large numbers of people if they should contaminate groundwater, relative to other types of impoundments. However, many impoundments are located near concentrations of people, and almost 87 percent are located over aquifers currently used as a source of drinking water (EPA, 1982b). About 50 percent are located over unsaturated and very permeable zones (EPA, 1982 b).

Contamination of groundwater by a particular impoundment will depend on soil permeability, depth to the water table, rates of evaporation and precipitation (including potential for overflow), geochemical characteristics of the soils (e. g., ion exchange and absorption), chemical composition and volume of the wastes, and other factors (EPA, 1978). For example, heavy metal movement depends on incorporation of the metals into

the bottom of the impoundments, leakage rates, and interactions of each metal with different underlying soils.

The contamination potential may be reduced if natural or artificial liners are located beneath the impoundment. The 1982 EPA survey indicated that only about 15-17 percent of all impoundments had liners, with a range of 10 percent for oil and gas impoundments to 28 percent for industrial impoundments. More recent data presented by EPA (*Inside EPA*, 1983d) indicate that 62 percent of all impoundments have at least a single liner; less than 22 percent have a double liner. In some States (e. g., California, Idaho, Illinois, Kentucky, Nevada, Oregon, and Pennsylvania) use of liners in all impoundment categories is widespread; in other States, use is widespread in only one or two impoundment categories.

EPA analyzed 416 case studies of groundwater contamination from impoundments and found that in 78.7 percent of the cases the contamination was caused by direct seepage, in 10.1 percent by dike failure or overflow, in 7.6 percent by liner failure, in 1.6 percent by catastrophic collapse, and in 2.0 percent by other causes. EPA also evaluated the impoundments’ potential to contaminate groundwater, water wells, and surface water as shown in table A.5. 1. Overall, 93 percent were judged to have intermediate or high potential for groundwater contamination.

8. and 9. Waste Tailings and Waste Piles

Mining operations generate two basic types of solid wastes—spoil piles and tailings. Spoil piles are generally disturbed soil and overburden from surface mining or waste rock from underground mining operations (Miller, 1980). Tailings are the solid wastes from the on-site operations of cleaning and extracting ores. Both types of solid wastes are often piled on the land surface or used as fill in topographic depressions confined by earthen dams (University of Oklahoma, 1983). They

Table A.5.1.—Contamination Potential of Surface Impoundments*

Impoundment category	High potential to contaminate groundwater	Potential to contaminate water wells	Potential to contaminate surface wells
Municipal	41 percent	27 percent	58 percent
Industrial	39 percent	29 percent	56 percent
Agricultural	26 percent	28 percent	61 percent
Mining	25 percent	17 percent	64 percent
Oil and gas	8 percent	17 percent	68 percent

*Data for “high potential to contaminate groundwater” are independent of data for other two columns.

SOURCE: EPA, 1982b.

are discussed together in this section because it is not always clear in the literature which source category is being referred to.

Development of Estimates of Numbers and Amounts

Metal and non-metal mines (excluding coal mines) produced 1.5 billion tons of waste rock in 1972 (EPA, 1977a); estimates of known amounts of tailings range from 215 million tons at both inactive and active uranium mining sites (Thomson, et al., 1983) to 250 million tons deposited in ponds annually by the metal mining industry (Miller, 1980). These figures total 1.72-1.75 billion tons, approximately 86 percent of which is in the form of waste piles (i. e., 1.5 billion tons of waste rock in 1.75 billion tons of waste material).

Approximately 2.3 billion tons of *total* waste material, including radioactive tailings, are generated annually by mining operations (EPA, 1981b; OTA, 1983a); this figure apparently includes both waste piles and tailings (both radioactive and non-radioactive). If the 86 percent figure is applied to the total of 2.3 billion tons, approximately 2.0 billion tons are in waste piles and 0.3 billion tons are in the form of tailings. The proportion of tailings may increase in the future; for example, the amount of active uranium mill tailings is projected to increase to 1.0- 1.9 billion tons by the year 2000 (Landa, 1980; also see *Radioactive Disposal Sites*, below).

Hazardous waste piles may also be generated by industrial operations. Hazardous waste piles at 174 facilities contained an estimated 0.39 billion gallons in 1981 (Dietz, et al., 1984). In view of the fact that these waste sites include only those regulated under Federal laws, the number of sites and amount of material probably represent the lower bounds.

Potential for Groundwater Contamination

In terms of their numbers, amounts of material, and nature of their contents, waste piles and tailings are among the major potential sources of groundwater contamination, especially from uranium, copper, and coal mining (Thomson, et al., 1983; Pye, et al., 1983; Johnson, 1983; Landa, 1980). Approximately one-third of active tailings piles have contaminated nearby shallow aquifers (EPA, 1983d).

Precipitation percolating through spoil piles and tailings carries soluble substances (e. g., arsenic, sulfuric acid, copper, selenium, and molybdenum) and radioactive wastes (e. g., isotopes of uranium, thorium, and radium, including radium-226 which has a half-life of 1,620 years) to the underlying water table (University of Oklahoma, 1983; Thomson, et al., 1983). Arsenic, selenium, lead, manganese, molybdenum, and vanadium

have been found in groundwater in seven States at distances of up to 1.5 miles from tailings piles and at concentrations above Federal or State limits (EPA, 1983e).

The most serious side-effects are associated with sulfide minerals (Koch, et al., 1982). Sulfuric acid is often generated from coal mining spoils by the oxidation of the sulfides in the coal; subsequent percolation into the water table results in acidic groundwater. Other minerals (e. g., lead, silver, zinc, molybdenum, nickel, and copper) are commonly found as sulfide ores; mining these minerals can also lead to the production of sulfuric acid (Koch, et al., 1982). In addition, the acid can dissolve other contaminants adsorbed on the soil into groundwater.

Impacts on groundwater quality depend on several factors: the location, size, and configuration of piles and tailings; the composition of piles and tailings; the climate (e. g., rate of precipitation); hydrogeological characteristics; and the control technology employed. Groundwater protection is not provided at many existing tailings disposal sites (Thomson, et al., 1983).

In some cases, certain factors can reduce the potential for groundwater contamination or the numbers of people affected. For example, many mining and smelting operations occur in arid or remote regions (e. g., for copper and uranium; EPA 1983e; Koch, et al., 1982; Thomson, et al., 1983). Low-grade ore piles (e.g., copper) can be subjected to controlled leaching and the runoff collected for reprocessing (Koch, et al., 1982). Further, a low pH is often rapidly neutralized as the flow leaves the tailings (Thomson, et al., 1983).

10. Materials Stockpiles

Development of Estimates of Numbers and Amounts

Very little information has been obtained regarding either the numbers or the amounts of materials in stockpiles in the United States. Approximately 3.4 billion tons of various materials (e. g., coal, sand and gravel, crushed stone, copper ore, iron ore, uranium ore, potash, titanium, phosphate rock, and gypsum) were produced in 1979 (Koch, et al., 1982). Stockpile size is probably proportional to production in most cases; however, data comparing production and stockpiles are available only for coal, iron ore, phosphate rock, titanium, and gypsum (Koch, et al., 1982). Stockpiles represent approximately 20-25 percent of production for coal, iron ore, and gypsum (annual production is more than 700 million tons of coal, more than 240 million tons of iron ore, and about 15 million tons of gypsum) and approximately 5-8 percent of production for phosphate rock

and titanium (annual production is about 191 million and 20 million tons, respectively).

For a preliminary estimate of the total volume held by materials stockpiles, assume that 20 percent of total materials production is stored in stockpiles. The choice of this percentage is based on an aggregation of the above percentages for the individual minerals and is weighted toward the higher figures because of the larger tonnages produced for those minerals. Given the total annual materials production of 3.4 billion tons, approximately 700 million tons per year are stockpiled. Reliability of the estimate is low but should be within an order of magnitude.

Some descriptive information is available for coal production and stockpiling. Approximately 780 million tons of coal were produced in 1979. Coal is stored outdoors primarily by electric utilities, coke plants, and industrial users; the average coal pile contained 95,000 metric tons and was 5.8 meters high. Coal stockpiles at utilities were estimated at 185 million tons in 1980 (Koch, et al., 1982). Substances present in coal piles include aluminum, iron, calcium, magnesium, sodium, potassium, manganese, sulfur, and phosphate, with trace amounts of arsenic, cadmium, mercury, lead, zinc, uranium, copper, and cobalt (Koch, et al., 1982).

Potential for Groundwater Contamination

Problems associated with materials stockpiles are much the same as those associated with waste piles and tailings (see *Waste Tailings and Waste Piles*, above); the major difference is that materials stockpiles are not wastes. But for all, the concern is the ultimate disposition of the soluble substances. Water percolating through stockpiles can carry soluble substances to the groundwater. Chemical reactions within coal piles, in particular, can produce sulfuric acid and ferric sulfate, which can then be carried down to the groundwater by precipitation percolating through the pile.

11. Graveyards

Decomposing bodies in graveyards produce fluids that can leak to underlying groundwater, especially if non-leakproof caskets are used.

The potential for graveyards to contaminate groundwater depends on several factors. Groundwater contamination is primarily a function of soils and depth to groundwater. Areas with high rainfall and high underlying water tables are most vulnerable to contamination from graveyards. Studies of individual cemeteries indicate that, in all cases, soil contamination occurred in immediate proximity to the graves but not all graveyards actually contaminated groundwater (Bouwer,

1978). Although the contamination potential cannot be accurately quantified, the magnitude of contamination appears to be highly localized and is probably much less than that from other sources.

12. Animal Burial

Animal burial procedures have become increasingly sophisticated. Mass burial—less common than individual burials—occurs near large concentrations of livestock and in local landfills or open dumps. Individual burials are most likely to take place within sections of municipal landfills or in residential backyards.

There are no data to assess the potential contribution of this source to groundwater contamination. It is highly site-specific and depends on disposal practices, the surface and subsurface hydrology, the proximity of the site to water sources, the nature and amount of the disposed material, and the cause of death.

13. Aboveground Storage Tanks

Aboveground storage tanks are used in industrial, commercial, and agricultural operations and at individual residences for a large variety of chemicals. No systematic information is available regarding numbers, sizes, and locations of these tanks or of the chemicals stored in them.

14. Underground Storage Tanks

Underground storage tanks are used by industries, commercial establishments, and individual residences for storage and treatment of products or raw materials, waste storage and treatment, and piping systems (San Francisco Bay Regional Water Quality Control Board, 1983; University of Oklahoma, 1983). Little information is available regarding treatment tanks; unless otherwise indicated, the discussion below refers to storage tanks. In addition, information about steel and fiberglass tanks will be distinguished whenever possible.

Industrial use is primarily for fuel storage but also for storage of a wide range of other substances including acids, metals, industrial solvents, technical grade chemicals, and chemical wastes (San Francisco Bay Regional Water Quality Control Board, 1983; California Assembly Office of Research, 1983). Commercial businesses (e. g., airports, corporations with car fleets, recyclers, farmers, and trucking industries) and individual homeowners use underground storage almost exclusively for fuel storage. Underground storage tanks are widespread throughout the country; gasoline storage tanks are concentrated in areas with high population density (and therefore with high automobile usage).

Development of Estimates of Numbers

The most numerous underground storage tanks are those used for gasoline at service stations and for fuel oil at residences. Based on the number of independent and major service stations in the United States (Lundberg, 1982) and on the average number of underground tanks per station, approximately 1.2 million *steel* underground tanks are found at service stations alone (Rogers, 1983).³ Approximately 100,000 *fiberglass* tanks also are used for underground storage of petroleum products and several thousand are used for non-petroleum products (Hammond, 1983).

Many other underground storage tanks, both known and unknown (and both active and abandoned), are used for petroleum and non-petroleum products throughout the country (Dalton, 1983; Rogers, 1983; White, 1983). The 1.2 million steel tanks at service stations may represent only one-fourth to one-third of the underground steel storage tanks for all products, the remainder being used by trucking companies, corporations, farmers, government agencies, and others (Rogers, 1983; White, 1983). White (1983) estimates that about 25 percent of all steel storage tanks are used by the petroleum industry (half of them by major producers and half by independent retailers), 25 percent by farmers, 5-6 percent by government agencies, and the remainder by various users. Note that the estimate that one-fourth to one-third of all steel underground tanks are used for petroleum may be too low for two major reasons. First, it seems to be based on data from Santa Clara County, CA, where the number of industrial chemical solvent storage tanks may be higher, and the relative number of tanks used for petroleum lower, than is typical of most of the country because of the number of high-technology industrial firms in Santa Clara County (Donovan, 1983). Second, approximately 60 percent of the 40,000 tanks produced annually for the last 5 years (28,000 steel and 12,000 fiberglass) have been installed at service stations (Donovan, 1983).

OTA'S study assumes that the number of steel tanks at service stations represents about one-half of all steel tanks. This figure is a compromise between the one-fourth to one-third and the 60 percent, weighted toward the latter because it is based on more reliable data. Using this assumption yields an estimate of 2.4 million steel underground tanks in the United States. The additional fiberglass tanks used for storing petroleum and non-petroleum products bring the total estimate to 2.5 million underground storage tanks for all non-hazardous products.

³This is a generally accepted figure and is cited by EPA (*Inside EPA*, 1983c) and by the Steel Tank Institute both in publications (e. g., Steel Tank Institute, 1983) and personal communications. Feliciano (1984) estimated that approximately 1.4 million underground tanks were used for storing gasoline.

There were at least 2,031 hazardous waste storage tanks and treatment tanks regulated under RCRA in 1981 (Dietz, et al., 1984); this figure does not include hazardous waste tanks operating under NPDES permits. Just how many of these are underground or above-ground is not known, but they are considered as an underground source in this analysis.

Development of Estimates of Amounts

It is very difficult to obtain an accurate estimate of the amount of material stored in underground storage tanks, but one approach involves using the average capacity of known tanks. The average service station underground steel tank held 4,000-6,000 gallons in the 1950s and now holds about 10,000 gallons; the largest registered steel tank has a capacity of 50,000 gallons (Donovan, 1983). The average capacity of fiberglass tanks is also about 10,000 gallons (Steel Tank Institute, 1983). Assuming an average 10,000-gallon capacity for underground tanks, the 2.5 million underground storage tanks have an estimated capacity of 25 billion gallons. The hazardous waste storage tanks and treatment tanks contain an estimated 13.8 billion gallons (Dietz, et al., 1984); this figure does not include hazardous wastewaters stored in tanks for less than 90 days or in tanks operated under NPDES permits.

Design, Operation, and Maintenance Characteristics

The installation and use of underground storage tanks are often not regulated. Most often the only regulations are local requirements for construction and installation, but even in these cases follow-up or periodic checks are rarely required to determine whether leaks have developed. Cathodic protection for steel tanks was seldom provided until recently; most tanks more than 15 years old are unprotected (Hammond, 1983).

There are no design requirements at the Federal level or in many States for storage facilities that might pose a threat to groundwater. At a minimum, design requirements should address (API, 1976): 1) tank construction—**e.g., to ensure compatibility with stored** substances and with local soil conditions; 2) reserve capacity; 3) safety devices—e. g., cutoff devices; and 4) inspection. The typical design life of tanks varies from 15-20 years for unprotected steel tanks and is highly dependent on environmental conditions. Leaks typically begin within 7 years of installation in humid areas or if tanks are in contact with salt-water, but they may not occur for more than 30 years in arid areas (Feliciano, 1984). No information was available about the typical design life of protected steel tanks but presumably it is more than 20 years. The design life of fiberglass tanks is estimated

at 40-50 years (Hammond, 1983); this figure is only a prediction—fiberglass tanks have been used commonly only since 1970, and the oldest one that has been tested for leaks is 13 years old.

The Pollution Liability Insurance Association no longer insures steel tanks more than 20 years old unless they meet stringent testing requirements (Morrison, 1983). Fiberglass tanks are warranted for up to 30 years (Hammond, 1983), but the Underwriters Laboratories insurance standards for fiberglass tanks do not cover alcohol blends (e. g., ethanol; Steel Tank Institute, 1983).

Potential for Groundwater Contamination

Underground storage tanks are known to have caused many cases of groundwater contamination (e. g., San Francisco Bay Regional Water Quality Control Board, 1983). In particular, old corroded gasoline storage tanks are frequently cited as sources of contamination (University of Oklahoma, 1983). As many as 77 percent of underground steel tanks may be affected by point corrosion (Rogers, no date). Such corrosion can be caused by impurities in the backfill, faulty installation involving surface abrasions and failure to remove shoring, and certain soil conditions (e. g., involving acidity, electrical resistance, presence or absence of sulfides, or moisture content).

Many companies have installed new tanks near old ones. When they do, a new tank often acts as a “sacrificial anode” (i. e., metallic ions flow from the new tank to the old tank) and it rusts faster (Dalton, 1983). In addition, dispensing pumps can develop leaks in couplings and hoses, and delivery lines can corrode or break (Dalton, 1983). Although new underground tanks are usually coated with a protective or corrosion-resistant material if they are steel or are made from relatively corrosion-resistant materials (e. g., fiberglass), they are still subject to corrosion-induced leakage. Fiberglass tanks can crack if installed incorrectly, and the polyester resins in fiberglass may be weakened by some alcohol-blend gasolines (Feliciano, 1984).

Tank age may be a principal factor in groundwater contamination (Rogers, 1983). Leaks have been observed in underground steel tanks aged 5-45 years but about one-third occur in tanks aged 15 years or less (Rogers, 1983). In New York, 60 percent of the leaks are in tanks older than 16 years, and 86 percent are in tanks more than 10 years old (New York State Department of Environmental Conservation, 1982). Many steel tanks in the United States are now in their mid-teens or older; the National Oil Jobbers Council estimates that nearly one-third are more than 16 years old (cited in Larson, 1983). Rogers (1983) directed a study

of 46,000 steel tanks owned by major oil and gas producers and found the following age composition: 4 percent less than 5 years, 20-23 percent between 5 and 10 years, 27 percent between 10 and 15 years, 21 percent between 15 and 20 years, and approximately 25 percent over 20 years. The age structure of this sample is probably younger than if a comparable sample had been taken from independent retailers because the major producers have recognized the potential for older tanks to leak and in the 1970s began to replace their older tanks (Donovan, 1983).

Rogers developed a model for predicting where leaks will occur, based on tank age and local soil conditions; it can also be used to estimate the number of leaking tanks. The leakage rate is assumed to increase as the tank population ages. Results from the model have been tested for approximately 10,000 tanks. Based on the age composition of the tanks and projected annual rates, Rogers estimated that about 50,000 tanks were leaking in 1982 and approximately 90,000-100,000 tanks would leak in 1983. This figure could be low because Rogers also estimated that approximately 25-30 percent of all steel tanks probably leak. If so, up to 720,000 underground steel tanks could be leaking (applying the upper figure of 30 percent to the 2.4 million steel tanks). EPA estimates that up to 240,000 tanks may be leaking and that the figure may increase to 75 percent of the total in the next 5 years (*Inside EPA*, 1983c).

Whether a leak contaminates groundwater is highly dependent on site-specific conditions including the concentration of the contaminant and the flow rate of the particular leak. For example, not all leaks at service stations contaminate groundwater. In fact, Rogers (1983) estimates that 85 percent of underground tank leaks at service stations do not go beyond the station boundary (because of the small amount of leakage or early detection) and do not contaminate groundwater; these incidents have typically cost \$20,000-\$30,000 to clean up. Another 10 percent of the leaks are estimated to travel beyond service station boundaries but are detected before they contaminate groundwater; typical costs of these operations are \$150,000. However, 5 percent of the leaks do contaminate groundwater, with typical cleanup costs of \$2.5-\$5 million and as high as \$11 million.

15. Containers

Containers are storage barrels and drums for various waste and non-waste products. They can be moved around with relative ease, and although they **may** be buried, they are not specifically designed to be. Very little information is available about containers because

they are not covered by any Federal water quality regulations. In 1981, about 3,577 facilities used containers for the storage of 0.16 billion gallons of hazardous wastes (Dietz, et al., 1984.) These figures are only for containers regulated under RCRA; actual numbers and amounts could be considerably higher.

16. Open Burning and Detonation Sites

Very little information is available on this source. Although there are probably many cases of waste materials burned in backyards or at landfills, these cases are classified here under the open dump, residential disposal, or landfill sources. Detonation sites are more structured (i.e., designed) operations; burning grounds could be either structured or unstructured. In 1981, 240 facilities regulated under RCRA incinerated 0.45 billion gallons of hazardous wastes (Dietz, et al., 1984).

The Department of Defense operates a number of burning grounds and ammunition detonation sites. Twelve such sites have been surveyed at Army installations, and TNT (and other hydrocarbons) and heavy metals (e. g., cadmium and chromium) have been detected in soil and in groundwater (U.S. Army Toxic and Hazardous Materials Agency, 1983). Several commercial and industrial sites listed on the National Priorities List by EPA (under CERCLA) have had fires or were operated as burning sites; groundwater contamination has been detected at all these sites.

17. Radioactive Disposal Sites

Radioactive materials arise from the nuclear fuel cycle, commercial and industrial products and wastes, and natural sources. They may have long half-lives, and they can migrate with no visible evidence. Natural radiation (e. g., radon-222) occurs throughout the United States, with the highest concentrations in granite formations (e.g., in Maine) and gypsum (e.g., in Florida).

Five basic types of waste products are produced in the development and generation of nuclear fuel and radioactive materials (DOE, 1983):

1. *Spent fuel* is the discharged irradiated fuel resulting from nuclear powerplant operations. It includes cesium-137 (half-life 28 years), strontium-90 (half-life 33 years), and cobalt-60 (half-life 6 years). Wastes containing these isotopes may need several hundred years or more to decay to low levels of radioactivity, with some estimates ranging as high as 100,000 years (University of Oklahoma, 1983).
2. High-level wastes are from the initial processing of irradiated reactor fuels. They are extremely ra-

dioactive, must be stored in specially constructed facilities, and eventually are either reprocessed or transferred to the Federal Government for long-term storage or permanent disposal (DOE, 1983).

3. *Transuranic wastes*, defined on the basis of specific radioactive criteria (DOE, 1983), result primarily from fuel reprocessing and from the manufacture of plutonium-containing products.
4. Low-level wastes are generated in liquid, gaseous, and solid forms and consist of a wide range of materials having generally low but potentially hazardous amounts of radiation (this category excludes uranium mill tailings). Low-level radioactive wastes are generated by nuclear reactors used for power production, weapons production, research (e.g., at universities and hospitals), and commercial products or activities (e. g., at hospitals). They can be in the form of discarded equipment, assorted refuse, and materials from decontamination facilities. They are either diluted until no longer classified as radioactive, disposed of indiscriminately, or shipped to approved low-level disposal sites.
5. *Uranium mill tailings* are the earthen residues left after the uranium is extracted from ores. Uranium refining also generates small amounts of solid, or semi-solid, low-level radioactive waste. Although the chemistry of the wastes varies among refineries, radium-226, thorium-230, and uranium-238 are usually present in small but significant concentrations. Disposal has commonly occurred in shallow burial grounds located near the refineries. (The waste rock associated with these radionuclides is discussed under Waste Tailings and Waste Piles, above).

Development of Estimates of Numbers

Prior to the mid- 1970s, low-level radioactive wastes were routinely packaged and shipped to commercial shallow nuclear waste burial sites. Six commercial sites were in operation, but three have been closed and two are accepting severely reduced volumes; the major remaining site is in the State of Washington. The Departments of Energy and Defense also maintain 22 sites for low-level waste disposal (DOE, 1983). High-level radioactive wastes are deposited at four regulated sites (Hanford, WA; Idaho Falls, ID; Aiken, SC; West Valley, NY) or are contained on-site at their place of generation (see OTA, 1982). Seven sites are used for transuranic waste disposal. Commercial spent fuel is usually stored at reactor sites or at two specific disposal sites.

Because different types of wastes are sometimes sent to the same site, the number of disposal sites is actually

less than the total of 38 in the above figures. Although recent legislation has called for State cooperation in site development for low-level radioactive waste disposal, commercial generators of low-level wastes are likely to be faced with possession of these wastes for some time. Remedial actions at inactive mill tailings sites are to be conducted by DOE under the Uranium Mill Tailings Radiation Control Act, but these actions have not yet begun (DOE, 1983; see ch. 9).

Development of Estimates of Radioactive Waste Production

A total of 4.80 million cubic yards of radioactive wastes was contained at various storage sites as of December 31, 1982 (DOE, 1983). This total was distributed as follows: 0.41 million cubic yards of high-level wastes, 0.48 million cubic yards of transuranic wastes, 3.78 million cubic yards of low-level wastes, and approximately 7,400 tons of spent fuel (the first three figures are based on DOE, 1983; the last on Hileman, 1982). Uranium mill tailings are discussed under *Waste Tailings and Waste Piles*, above.

Potential for Groundwater Contamination

Radioactivity is a major threat to groundwater because of the longevity of isotopes and their ability to migrate unnoticed. Much debate centers on the efficacy of waste disposal burial methods over time; for example, disposal containers are often deposited in or above shallow water tables. Some isotopes enter groundwater from radioactive wastes, but other isotopes are present because of the leaching of natural geologic substances (e. g., gypsum). It is estimated that 10-30 square miles of land are underlain by groundwater contaminated beyond potable use by radioactive wastes (USGS, 1983).

Numerous radionuclides have been detected in groundwater as shown in table A.5. 2. These radionuclides emit three types of radiation: alpha (a), beta (~), and gamma (T) (League of Women voters Education Fund, 1980). Alpha radiation has the least power to penetrate skin, but it can cause severe tissue and organ damage if it enters the body through ingestion of contaminated drinking water or food or through inhalation. Beta radiation is more penetrating, but it also is most serious when ingested or inhaled. Gamma radiation has the greatest power to penetrate skin and usually is associated with beta radiation; it too can damage critical organs.

18. Pipelines

Pipelines are used to transport, collect, and/or distribute both wastes and non-waste products. The wastes are primarily municipal sewage, most often located in

Table A.5.2.—Categorization of Known and Potential Radionuclides in Groundwater by Mode of Decay

Radio nuclide ^a	~	p	@ and y, combined	y
Antimony-125.....			X	
Barium-140.....			X	
Cesium-134.....			X	
● Cesium-137.....			X	
● Chromium-51.....				X
● Cobalt-60.....			X	
iodine-129.....			X	
● iodine-131.....			X	
● iron-59.....			X	
● Lead-210.....			X	
● Phosphorus-32.....		X		
● Plutonium-238.....	X			
● Plutonium-243.....			X	
● Radium-226.....	X			
● Radium-228.....		X		
Ruthenium-103.....			X	
● Ruthenium-106.....		X		
● Scandium-46.....			X	
Strontium-89.....		X		
● Strontium-90.....		X		
Strontium-131.....			X	
● Thorium-270.....	X			
● Tritium.....		X		
Uranium-230.....	X			
● Uranium-238.....	X			
● Zinc-65.....			X	
● Zirconium-95.....			X	

^aRadionuclides marked with an asterisk are known to have contaminated groundwater and are documented by at least two of the listed sources. Alpha (α), beta (β), and gamma (γ) radiation are discussed in the text.

SOURCE Woodward-Clyde Consultants, Inc., 1983; University of Oklahoma, 1983, Environ Corp., 1983.

densely populated areas. The primary non-wastes are petroleum products and natural gas, but ammonia, coal, sulfur, and anhydrous ammonia are also transported (University of Oklahoma, 1983). Non-waste pipelines are located throughout the Nation; maps of major pipeline networks are available from the Federal Energy Administration (University of Oklahoma, 1983).

Development of Estimates of Numbers and Amounts

Approximately 175,000 miles of pipeline carrying 9.63 billion bbls of petroleum products per year were in operation in the United States in 1976 (Pye, et al., 1983). Information presented in Miller (1980) indicates that approximately 700,000 miles of sewer pipeline were in use in 1980. In 1978, 154 million people were served by sewer pipelines (U.S. Department of Commerce, 1981). Assuming an average sewage flow of 100 gallons per day per person (Miller, 1980), approximately 5.6 trillion gallons of sewage were transported by sewer pipelines in 1978.

Potential for Groundwater Contamination

Although pipelines are designed to retain their contents and thus pose no threat to groundwater, in reality they have a contamination potential through leakage. The major causes of leaks are ruptures, external and internal corrosion, incorrect operating procedures, and use of substances such as paint products, battery fluids, defective welds or pipes. In 1981, these causes accounted for 41 percent, 22 percent, 7 percent, and 6 percent of all reported leaks, respectively (DOT, 1981). Other causes were surges (e. g., floods) of fluid in pipelines, breakage or heaving of lines by tree roots, earthquakes, loss of foundation support, and rupture due to other loads. Miller (1980) estimated that leakage from sewer pipelines was around 5 percent; if it is, approximately 280 billion gallons of sewage annually could be leaching into groundwater. This estimate of leakage is based on the unverified assumption of 5 percent leakage.

Because interstate pipelines are a major means of transporting materials, they are regulated by the Department of Transportation (DOT); and any leaks and spills must be reported to DOT (see ch. 3 and app. B. 1). However, collection and distribution systems, gas stations, residential users, and even relatively large intrastate carriers are not required to report leaks and spills. Collection and distribution pipelines are not regulated other than during their initial installation to prevent the escape of combustible, explosive, or toxic chemicals; the potential for groundwater contamination is not a primary consideration.

About 4,100 non-waste liquid pipeline leaks and accidents were reported from 1968 through 1981 (DOT, 1981; the figure is not certain because information differs on pp. 21 and 39). Of that number, 2,813 occurred from 1971-81, with 3.4 million bbls of material lost. In 1981, 239 pipeline failures were reported, with 214,384 bbls lost; various products were involved in the leaks: crude oil was involved in 48.1 percent of the failures, gasoline in 19.3 percent, liquified petroleum gas (LPG) in 14.6 percent, natural gas liquid (LNG) in 5.0 percent, and fuel oil in 4.6 percent. The remaining materials involved were jet fuel, diesel fuel, anhydrous ammonia, kerosene, turbine fuel, oil and gas, and condensate.

19. *Material Transport and Transfer Operations*

Material transport and transfer operations refer to the movement of substances by vehicle (e. g., truck and railroad) along transportation corridors. Handling facilities such as airports and loading docks are also included.

Development of Estimates of Numbers and Amounts

Estimates of the number of spills vary. The National Academy of Sciences (NAS) estimated that approximately 16,000 spills occur annually, involving a variety of substances such as paint products, battery fluids, gasoline, corrosive compounds, flammable compounds, various acids, and anhydrous ammonia (NAS, 1983 b). The Council on Environmental Quality (CEQ) (1982) reported on 10,072 known spills of oil or hazardous chemicals totaling 19.6 million gallons in 1981; however, these spills include leaks from storage pipelines and as well as from transportation facilities. DOT reported 9,063 incidents involving hazardous materials in 1981 and 6,540 in 1982 (as of Apr. 30, 1983; Jossi, 1983). Almost 81 percent of the 1982 incidents involved commercial carriers on highways, another 5 percent involved private carriers on highways, 13 percent involved railways, and the remainder involved other forms of transportation.

Very little information was available about the amount of hazardous materials lost in spills, other than the CEQ figure cited above; and no information was available regarding non-hazardous materials. NAS estimated that about one-half of the 4 billion tons of hazardous materials transported annually in the United States is transported on highways (NAS, 1983 b). EPA (Inside EPA, 1983c) estimated that about 90 percent of all transportation of hazardous wastes is by truck. Further, EPA also estimated that when hazardous materials are transported by truck, approximately 0.35 percent of the hazardous materials (slightly more than 38 gallons) are lost during each shipment of 200 55-gallon drums. Assuming that the same 0.35 percent loss rate applies to the entire 4 billion tons shipped annually in the United States, no matter how transported, approximately 14 million tons of hazardous materials are spilled during material transport and transfer operations. This estimate is only a first approximation.

Potential for Groundwater Contamination

Transport and transfer of materials have the potential to contaminate groundwater contamination through spills and leaks. Spills are generally unintentional and can occur at random at transport facilities and along transportation corridors. Although an estimate can be developed for the amount of material spilled annually (see above), it is not possible to estimate the amount of spilled material that threatens groundwater.

Storage and transfer facilities for oil and hazardous chemicals must be designed and certified by a registered

engineer if they pose a threat to surface water (University of Oklahoma, 1983). However, similar design requirements do not exist at the Federal level or in many States for groundwater (University of Oklahoma, 1983; see app. H.3). Design procedures that would take into account the potential for groundwater contamination relate to (API, 1976): drainage systems at loading and unloading areas, containment systems for possible spills, security measures, and tanker/tank design and interface.

20. Irrigation Practices

Water used for irrigation tends to percolate into the subsurface and move toward discharge points. As it does, it carries with it substances applied to and associated with the soil (e. g., fertilizers, pesticides, and sediment).

Development of Estimates of Numbers and Amounts

About 14 percent of cropland in the United States is irrigated; 58 million acres were irrigated in 1977 (USDA, 1981a), and 51 million acres were irrigated in 1978 (U.S. Department of Commerce, 1982). Irrigation is most common in the West, the Central and Southern Plains, Arkansas, and Florida (U.S. Department of Commerce, 1982). Approximately 169 million acre-feet of water were used for irrigation in 1980 (CEQ 1982; the figure includes both surface water and groundwater). About 68 percent of the total groundwater use in 1980 was for irrigation (USGS, 1984).

Potential for Groundwater Contamination

Although salts, pesticides, and fertilizers may be present wherever crops are grown, irrigation return flows tend to concentrate these chemicals (University of Oklahoma, 1983) and can reduce agricultural productivity. Groundwater salinity (i.e., dissolved salts) can increase because of evaporation, transpiration, and subsequent leaching of saline soils. Irrigation practices have increased groundwater salinity in many parts of the West and Southwest (Sheridan, 1981).

Data are lacking about the proportion of irrigation water that is consumed by crops, percolates into the subsurface, and runs off the land. Salinity is difficult to reduce because the volume of irrigation water is difficult to alter and because much of the salt in water occurs naturally. However, various water conservation practices and the application of more efficient irrigation technology can decrease salinity significantly (USDA, 1981b; OTA, 1983 b).

21. Pesticide Applications

Pesticides are chemicals used for control of insects, fungi, and other undesirable organisms and weeds. Agricultural operations (including but not limited to those on irrigated lands) account for most pesticide use (69-72 percent), government agencies and industrial/commercial organizations account for 21 percent, and home and garden uses account for the remainder (EPA, 1980a; Seiber, 1981).

Pesticide Production and Estimates of Use

Approximately 1.4- 1.5 billion pounds of pesticides are produced in the United States each year (USDA, 1983a; EPA, 1977b; Forest Pest Management Institute, 1982). Production has doubled since the mid-1960s (EPA, 1980e) and is growing approximately 1.4 percent annually (Forest Pest Management Institute, 1982). Pesticides are composed of 1,200-1,400 active ingredients in approximately 2,500 intermediate products; these products in turn are formulated into some 50,000 registered end-use pesticide products (Roelofs, 1983; EPA, 1977 b). Depending on the definition, there are approximately 30-80 major pesticide manufacturers, 100 smaller producers, 3,300 formulators, and 29,000 distributors in the United States (EPA, 1980a; USDA, 1983a).

Of the 1.43 billion pounds of end-use products manufactured in 1981, 839 million pounds were herbicides, 448 million pounds were insecticides, and 143 million pounds were fungicides. In 1982, it is estimated that 57.8 percent of the herbicides were amides and triazines and that 69.9 percent of the insecticides were organophosphates (Schaub, 1983).

Use of pesticides on cropland can be measured by the pounds of active ingredients applied and by the number of acre-treatments (i. e., the number of acres treated, including acres treated more than once). Approximately 552 million pounds of *active ingredients* were applied to major field crops in 1982 (USDA, 1983c)—451 million pounds of herbicides, 71 million pounds of insecticides, and 30 million pounds of fungicides, fumigants, dessicants, defoliant, growth regulators, and miticides. Pesticide applications may average as much as 2.6 pounds per acre (USDA, 1981a); in 1976, 2.2 pounds of insecticides and 2.0 pounds of herbicides were applied per acre (CEQ 1982). However, new products have been developed which require as little as 0.1 pound of active ingredients per acre (Schaub, 1983); some new chemicals may require even less (Kearney, 1983).

Approximately 280 million acre-treatments are conducted annually (Schaub, 1983; USDA, 1978). The four major crops—corn, cotton, soybeans, and wheat—

account for 85 percent of all herbicide use and 70 percent of all insecticide use (Eichers, 1981). Forty-seven percent of all insecticides are applied to cotton (USDA, 1981a). About 85-90 percent of the corn, cotton, soybean, and rice acreage is treated with herbicides.

Airplane applications accounted for 65 percent of all pesticide applications on agricultural and forest lands in 1978 (USDA, 1978). These applications involved some 10,000 aircraft treating more than 180 million acres (Kearney, 1983).

Potential for Groundwater Contamination

Groundwater contamination from the use of pesticides in agricultural operations has been found in at least 18 States (Cohen, et al., 1984; Rothschild, et al., 1982; Spalding, et al., 1980); at least 12 different pesticides were involved (Cohen, et al., 1984). Contamination can occur from common use practices, spills, accidents, disposal of excess pesticides, disposal of wastewater from equipment and from rinsing empty containers, and other causes (Hall, 1983; *Chemical and Engineering News*, 1983). Contamination potential can generally be reduced through methods of use, storage, and disposal (*Chemical and Engineering News*, 1983).

However, airplane applications pose special problems. The disposal of wastewater from airplanes (either before or after landing) is often haphazard and may take place in ditches, lagoons, streams, and sewers or on the land (Seiber, 1981). It is estimated that the operation of one plane results in approximately 10,000 gallons of wastewater and 44 pounds of pesticides that must be disposed of each year (Seiber, 1981). Given the 10,000 aircraft involved, approximately 100 million gallons of wastewater and 440,000 pounds of pesticides must be disposed of annually.

Movement of pesticides through soil and into groundwater depends on a variety of pesticide-specific and site-specific factors including water volubility, vapor pressure, speciation, hydrolysis half-life, photolysis half-life, soil/water adsorption coefficient, depth to the water table, soil type, and rainfall (Cohen, et al., 1984; Severn, et al., 1983). Severn, et al. (1983) list quantitative conditions under which groundwater contamination can occur.

Many compounds do not move much with actual groundwater flow but adhere to and move with the soil particles themselves (e. g., many hydrocarbons; Hall, 1983). Other compounds are more soluble and move relatively rapidly (e. g., Temik or aldicarb; Hall, 1983); these compounds pose problems, especially in areas with high water tables (e. g., Florida). USDA is conducting at least 37 projects on the movement and fate of pesticides in the soil (Helling, 1983; also see ch. 3).

22. Fertilizer Applications

Farmers used 54.0 million tons of commercial fertilizers in 1980-81, 48.7 million tons in 1981-82, and 42.3 million tons in 1982-83 (USDA, 1983d). The areas covered are likely the same as those covered by pesticides and are spread throughout much of the country (University of Oklahoma, 1983; USDA, 1982a); the five States using the most fertilizer in both 1981-82 and 1982-83 were Illinois, Iowa, California, Indiana, and Texas (USDA, 1983d). Fertilizers used in 1981-82 contained 11.1 million tons of nitrogen (22.8 percent of the total 48.7 million tons), 4.8 million tons of phosphates (9.9 percent), and 5.6 million tons of potash (1 1.5 percent) (USDA, 1983d). The USDA has estimated that nutrient application rates range from 0.03-8.4 pounds per acre for nitrogen and from 0.01-0.08 pounds per acre for phosphorus (USDA, 1981 b). In 1978, approximately 229 million acres were treated with commercial fertilizers and 17 million acres were treated with lime (U.S. Department of Commerce, 1982).

The potential for fertilizers to contaminate groundwater depends on the rate of application in relation to crop uptake (University of Oklahoma, 1983). This rate is often difficult to control because farmers generally apply enough fertilizer for the entire growing season prior to planting (Swanson, 1983).

23. Animal Feeding Operations

In the last two decades the number of animal feedlots with more than 1,000 animals has increased rapidly (Miller, 1980). In 1982, there were 1,935 cattle feedlots in the United States marketing approximately 16.8 million cattle; 969 of the feedlots, with a capacity of more than 2,000, marketed 15.3 million cattle (USDA, 1983 b). The feedlots are located primarily in the Corn Belt and High Plains. Inventories of animals on farms and feedlots during 1978 showed a total of 106 million cattle and calves, 59 million hogs and pigs (USDA, 1982 b), 12 million sheep and lambs, 2.2 million horses and ponies, more than 359 million chickens, and more than 140 million turkeys (U.S. Department of Commerce, 1982). The principal rearing region is the South for poultry, the West for sheep, and the Midwest for hogs.

Estimates of Manure Production

Cattle are estimated to produce 0.5 tons of manure during their 4-5 month stay in feedlots (Pye, et al., 1983). Thus in the larger cattle feedlots (i.e., with more than 1,000 animals), more than 8 million tons of manure

are produced annually. The USDA has estimated that all livestock on feedlots and farms produce 175 million dry tons of manure annually, and 90 percent of it is returned to the land (USDA, 1981a).

Potential for Groundwater Contamination

Animal feeding operations can adversely affect groundwater if leachate enters the subsurface either directly from the feedlots or from waste piles and wastewater impoundments (see *Surface Impoundments*, above). The most important potential contaminant in manure is nitrogen, but bacteria, viruses, and phosphates are also of concern (University of Oklahoma, 1983).

The potential for groundwater contamination is greatest in areas with high densities of animals and a shallow water table. Thus even small farms have the potential to contaminate groundwater; large numbers of animals in a small area can stress the natural assimilative capacity of the soil (Pye, et al., 1983). Of the 718,000 farms with fewer than 300 animals, 25 percent are estimated to have the potential to degrade water quality (USDA, 1981 b). Data are insufficient to estimate the volume of leachate and runoff that actually reaches the water table from large feedlots. In any case, because manure piles and feedlots often are near rural homes, domestic water supply wells are vulnerable.

24. De-Icing Salts Applications

Highway de-icing salts are applied to snow and ice-covered roads to improve driving conditions. The salts consist mostly of commercial rock and marine salt, with the addition of ferric ferrocyanide and sodium ferrocyanide to minimize caking of the salts when stored; other additives include chromate and phosphate, which reduce the corrosiveness of the salts (Bouwer, 1978). Use of highway de-icing salts is confined primarily to the *snowbelt*, especially the populous areas of the Northeast and Mideast, and is dependent on weather conditions.

Development of Estimates of Numbers and Amounts

During the winter of 1982-83, a minimum of 9.35 million tons of dry salts and abrasives and 1.78 million gallons of liquid salts were applied to highways (Salt Institute, 1983; data were for agencies using more than 10,000 tons of total materials annually). More than 12 million tons of salt were used in the 1978-79 winter (Pye, et al., 1983).

Highway salting rates generally range from 355-1,065 pounds per mile (100-300 kilograms (kg) per kilometer) per application. During the course of a winter

season, roads typically receive 17.6 tons (16,000 kg) of salt per lane per mile, or approximately 88 tons (80,000 kg) per mile for a typical highway with four lanes and shoulders (Bouwer, 1978); this figure varies geographically and from year to year. During the 1982-83 winter, an average of 15.5 tons of dry salts and abrasives and 2.9 gallons of liquid salts were applied per lane per mile (based on Salt Institute, 1983).

Potential for Groundwater Contamination

Estimates of the total use of de-icing salts should be interpreted cautiously when attempting to assess their contribution to groundwater contamination. Although all salts used have the potential for reaching groundwater, the amounts likely to reach groundwater are unknown and depend on hydrogeological and other factors (University of Oklahoma, 1983).

Many cases of contamination caused by highway de-icing salts have been documented in snowbelt areas (Bouwer, 1978; Dalton, 1983; Lord, 1983). The sources are both the leachate from stockpiles of salt and the runoff from the roads. Major problems are primarily associated with the storage of salt (Lord, 1983); salt stockpiles are maintained year-round and are often entirely exposed.

Chloride levels in road runoff during snowmelt have been observed to range from 1,130-25,100 parts per million (Bouwer, 1978); drinking water is generally considered contaminated when chloride levels exceed 250 parts per million (NAS, 1980). Sodium ferrocyanide is soluble in water and, when exposed to sunlight, can generate cyanide in concentrations in excess of maximum drinking water limits (see app. C.3). Chromate additives can produce excessive concentrations of hexavalent chromium in meltwater (Bouwer, 1978).

Technology is now available to minimize leaching from salt stockpiles, but most research is being focused on what happens after application of de-icing salts (Lord, 1983). For example, the potential for groundwater contamination after application can be reduced by designing roads that require less de-icing and by collecting and disposing of the runoff, by developing substitute highway materials for maintaining safe driving conditions, and by developing alternatives to the de-icing salts now used.

25. Urban Runoff

Urbanization necessarily expands the areas that are impervious to rainfall and thus increases the amount and rate of surface runoff. The runoff, in turn, is channeled by extensive drainage networks and carries with it the contaminants associated with urban activities (e. g.,

automobile emissions, litter, deposited atmospheric pollutants, and sediments; University of Oklahoma, 1983). Any stormwater that infiltrates the surface can also carry these contaminants.

According to EPA (1983 c), over 21.2 million urban acres contributed stormwater runoff in 1970, and this figure is projected to increase to 32.6 million acres by the year 2000. Data are insufficient to determine the extent to which urban runoff and infiltrating stormwater contribute to groundwater contamination.

Potential for Groundwater Contamination

Urban runoff is a primary cause of degraded surface water quality in heavily populated areas. After flowing into existing water bodies, contaminants originally carried in runoff may accumulate in solution or in sediments (Owe, et al., 1982). The potential for groundwater contamination from urban runoff will depend on where the runoff is discharged, its proximity to aquifers, and various hydrogeologic factors.

A major source of contaminants is automobile emissions, which may contribute contaminants to surface runoff in some areas. The contaminants of most concern are suspended solids and toxic substances, especially heavy metals and hydrocarbons. Runoff can also contain bacteria, nutrients, and other oxygen-demanding loads, and petroleum residues (USDA, 1981a; Owe, et al., 1982). Contaminant levels in urban runoff are often higher than established ambient levels for receiving waters (Owe, et al., 1982).

26. *Percolation of Atmospheric Pollutants*

Many potential contaminants of groundwater are carried in the atmosphere and eventually reach the land surface through either dry deposition between storms or transport in water and snow during storms (Owe, et al., 1982). A number of sources of atmospheric pollutants are known, among them automobile emissions and various industrial processes. The major contaminants are sulfur and nitrogen compounds, asbestos, and heavy metals (Owe, et al., 1982). Their ultimate distribution depends on their size when they are released and on weather patterns while they are moving in the atmosphere.

Percolation of atmospheric pollutants into groundwater is greatest in areas of high air pollution. One of the better-studied cases involves acid rain. Although widely distributed, acid rain occurs predominantly around the Great Lakes, the Northeast, and south-central Canada (OTA, 1984).

27. *Mining and Mine Drainage*

Minerals are extracted by either underground mining or surface mining. Underground mining is used to extract deep, relatively high-grade ore from structurally stable rock. The methods used (e. g., room-and-pillar, block caving, and stoping; NAS, 1979) depend on topography, geology, and characteristics of the ore (e. g., size, shape, depth, and ore grade). In surface mining, pits are created when the overburden and topsoil are removed to expose large, shallow deposits (generally covered with less than 300 feet of loose soil; NRC, 1983); operations include quarrying, open-pit, opencut, open-cast, stripping, placering, and dredging (NAS, 1979). Deep underground mines, especially for coal, are located primarily in the Appalachian region; and surface mines are primarily in the West and Midwest.

Development of Estimates of Numbers and Amounts

More than 15,000 mines were in operation in 1976 (NAS, 1979). Wirries, et al. (1983) estimate that there are also 67,000 inactive or abandoned mines in the conterminous United States, 49,000 of them in the Midwest and Appalachia. The total land area that has been disturbed has been estimated at 4 million acres; the rate of disturbance may have been as high as 5,000 acres per week in the early 1970s (NAS, 1979). Approximately 383,000 acres have been abandoned.

Miller (1980) estimated that 3.6 million tons of acid were generated annually from the 200,000 acres used for the disposal of coal mining wastes (27,000 of those acres had been reclaimed). Depending on how many of the approximately 383,000 abandoned acres are also used for waste disposal, the amount of acid generated annually could be as high as 10 million tons (the additional acreage triples the total acreage and presumably the subsequent estimate). Miller (1980) also estimated that 10 percent of the acid generated enters groundwater; thus 0.36- 1.0 million tons of acid could enter groundwater each year.

Potential for Groundwater Contamination

Excavation and *operation* of both surface and underground mines can disrupt the natural positioning of aquifers and hence groundwater flow. As a consequence, water can percolate through the fractured overburden and mix with mine wastes and other materials that were previously separated (NRC, 1983; EPA, 1981a). The problem can be minimized by dewatering (e. g., pumping water to the surface, possibly at rates of up to 200-3,000 gallons per minute; NRC, 1983).

The primary problem concerning groundwater relates to the disposal of spent mill tailings, especially in underground mining. Underground mining introduces oxygen and water, which can result in the oxidation of pyrite and the subsequent formation of acid mine drainage—an acidic mixture of iron salts, other salts, and sulfuric acid (Thomson, et al., 1983). Acid mine drainage is a major problem in the East; in the West, groundwater seldom becomes acidic, usually because carbonates in the overburden help neutralize any acid produced. However, sulfate concentrations are often very high in Western surface mined lands. **Arsenic**, molybdenum, vanadium, and other minerals also can become soluble in the oxidizing conditions of mining in general and can enter groundwater.

Wirries, et al. (1983) studied inactive deep underground coal mines in Appalachia and the Midwest. Drainage quality was highly variable, with most sites exceeding Federal effluent guidelines. Trace metals (e.g., cadmium, **mercury**, zinc, and nickel) were present in low concentrations. Calcareous material in the overburden helped buffer acid drainage. The amount and rate of acid formation and the chemical quality of the drainage tend to be functions of the amount and type of pyrite present, characteristics of the overburden, and the amounts of air and water available for chemical reactions (EPA, 1981 a).

28. Production Wells

A variety of wells are included as production wells—oil, geothermal and heat recovery, and water supply wells. Oil wells are clustered in the Southwest, Alaska, Louisiana, Wyoming, and the Midwest. Geothermal activities are primarily in the West and in the heavily populated northern States where the use of earth-coupled heat pumps is increasing (University of Oklahoma, 1983). No comprehensive information on the location of water supply wells was collected as part of this study, but they are likely to be most numerous in areas with high groundwater withdrawals (the Southwest, the Central Plains, Idaho, and Florida; see Solley, et al., 1983).

Approximately 548,000 oil wells produced an estimated 3.1 billion bbls of crude oil in 1980 (U.S. Department of Commerce, 1981); the brine associated with these wells is discussed in *Injection Wells*, above. Abandoned production wells may number around 1.2 million (Kaplan, et al., 1983).

More than 376,000 irrigation wells are used to supply water for approximately 126,000 farms in the United States (*The Groundwater Newsletter*, 1983 b).

All production wells share a similar potential to contaminate groundwater. It is related to installation and

operation methods (e. g., for oil wells, the use of treatment chemicals, drilling fluids, and other chemicals), incorrectly plugged or abandoned wells, cross-contamination, and overdraft. Corrosion of screens and casings in unrepaired or abandoned wells can result in the wells becoming conduits for the vertical migration of contaminants (Gass, et al., 1977; see *Injection Wells*, above, for discussion of groundwater contamination problems associated with wells).

29. Other Wells

Other wells include those used in various monitoring and exploration activities. No systematic information is available regarding numbers and locations of these wells.

30. Construction Excavation

Excavation at construction sites has many purposes including: clearing, pest control, rough grading, facility construction, and the restoration of staging and stockpile areas upon completion of a job (University of Oklahoma, 1983). Construction excavation is intense in areas experiencing growth, but it is usually temporary.

Almost no data are available on the amount of materials that is excavated annually. It has been estimated that 45 million tons of junked auto, construction, and demolition wastes are generated annually (EPA, 1981b) but how much of these wastes results from construction excavation is not known.

Excavation at construction sites can produce potential groundwater contaminants in a variety of ways. Clearing and grubbing and pest control practices can produce contaminants from the use of pesticides and the decay of cleared vegetation. Heavy construction equipment used for rough grading can spill diesel fuel, oil, and lubricants. Some construction activities can include dust control in which oil, calcium chloride, and water are used. The concrete used in construction is a source of contaminants from washing, spills, and wastes (University of Oklahoma, 1983).

31. Groundwater—Surface Water Interactions

When groundwater aquifers are hydrologically connected with surface water, the aquifer can be partially recharged by infiltration of the surface water.⁴ If the sur-

⁴Alternatively, groundwater may replenish surface water, e. g., it may provide the baseflow for streams and rivers. In this case, contaminants in groundwater could be transferred to surface water

face water is contaminated, or if it reacts chemically with the subsurface materials as it infiltrates downward, degradation of groundwater quality can follow (Miller, 1980).

32. Natural Leaching

Natural leaching occurs on a local scale in aquifers, or in portions of aquifers, whose geologic materials can be dissolved into solution. No systematic information is available about the significance of natural leaching to groundwater contamination.

33. Salt-Water Intrusion/ Brackish Water Upconing

Approximately 21 billion gallons of groundwater per day—26 percent of all groundwater withdrawn (USDA, 1981a)—are withdrawn in excess of recharge capabilities (i. e., overdrafting, overpumping, or overmining). Withdrawals significantly in excess of natural recharge are located predominantly in coastal areas (e. g., California, Texas, Louisiana, Florida, and New York), the Southwest, and the Central Plains (USDA, 1981 b).

Overdrafting can disrupt the natural hydrologic processes associated with groundwater; and subsequent impacts on aquifers and groundwater quality include: salt-water intrusion in coastal areas, brine-water intrusion (or brackish water upconing) in inland areas, and intensified natural leaching. Land subsidence may also result; it disrupts the natural positioning of aquifers and has additional surface impacts (e. g., subsidence). Salt-water or brine-water intrusion is probably the major problem associated with overdraft but it occurs only in areas where freshwater aquifers are underlain by salt-water or brine. At some coastal areas, injection of freshwater into aquifers is used to prevent salt-water intrusion (University of Oklahoma, 1983).

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- 29 *Other Wells*
- 30 *Construction Excavation*
- 31 *Groundwater—Surface Water Interactions*
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