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Agricultural Drainage Wells: Impact on Ground Water

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Introduction

As part of the Safe Drinking Water Act (SDWA) enacted in 1974, Congress mandated the development of a Federally approved Underground Injection Control (UIC) program for each State, Possession, and Territory. The aim of the program was to prevent contamination of Underground Sources of Drinking Water (USDW) by injection wells. A well is defined in Title 40 of the Code of Federal Regulations as either a dug hole or a bored, drilled or driven shaft whose depth is greater than its largest surface dimension. Injection is defined as the subsurface emplacement of fluids in a well where a fluid is any material that flows or moves whether it is semisolid, liquid, sludge or gas.

Under the UIC program, five classes of injection wells are recognized. These are:

Class I: Wells used to inject hazardous wastes or dispose of industrial and municipal fluids beneath the lowermost USDW.

Class II: Wells used to inject fluids associated with the production of oil and natural gas or fluids/ compounds used for enhanced hydrocarbon recovery.

Class III: Wells which inject fluids for the extraction of minerals.

Class IV: Wells which dispose of hazardous or radio-active wastes into or above a USDW. (These wells are now banned.)

Class V: Wells not included in the other classes, generally inject nonhazardous fluid into or above a USDW.

Seven main categories of Class V injection wells consisting of over 30 individual well types are recognized under the UIC program. The well types range in complexity from simple shallow cesspools to sophisticated geothermal reinjection wells which may be thousands of feet deep. USEPA and State records show that at least 170,000 Class V wells are in existence in the United States and its Territories and Possessions of which about 57 percent are drainage wells and 26 percent are sewage related wells. Conceivably, more than one million Class V wells may be in existence. According to the EPA (1989), of the five classes of wells recognized under the UIC program, Class V wells may pose the greatest environmental threat to the Nation's ground-water resources.

What are agricultural drainage wells?

Agricultural drainage wells are one of many Class V well types which may pose a high potential for ground water contamination. Agricultural drainage wells are currently among 15 Class V well types which have been designated "high priority" by the USEPA under a preliminary screening program conducted by the Office of Drinking Water.

Agricultural drainage wells can be defined as constructed subsurface disposal systems used to accelerate the drainage of agricultural surface runoff and/or subsurface flow. Accelerated drainage is required in many regions of the United States in order to provide a well-aerated root zone for optimum crop growth. In Iowa, for example, accelerated drainage is necessary in many areas for optimizing row-crop production. Glanville (1985) cites an example in northcentral Iowa where about five million gallons of excess water must be drained annually from a flat 40-acre field.

Generally, an agricultural drainage well system consists of a buried collection basin or cistern, one or more tile lines entering the cistern, and a drilled, or dug, cased well (Report to Congress, 1987). Agricultural drainage wells may receive field drainage from precipitation, snowmelt, and floodwaters; irrigation return flow; and animal yard, feedlot, or dairy runoff. In drier regions such as the western United States, irrigation return flow is a principal component of flow entering agricultural drainage wells. In wetter regions characterized by poor soil drainage, field drainage is likely to be a principal component. Both irrigation return flow and field drainage can take the form of surface and subsurface runoff. Figure 1 shows a typical agricultural drainage well system for collection of both surface and subsurface flow.

Location of agricultural drainage wells

Generally, agricultural drainage wells may be found in areas having low soil permeabilities, shallow water tables, and insufficient natural surface drainage (Report to Congress, 1987). According to an inventory of agricultural drainage wells reported in the 1987 Report to Congress, 1,338 such wells have been identified throughout the United States. Of those wells identified, the majority were confined to just a few states with the greatest number of wells being identified in Idaho (572), Iowa (230), New York (150), Texas (108), and Indiana (72). A list of States in which the presence of agricultural drainage wells has been confirmed is provided in Table 1. The difficulties and uncertainties inherent in compiling the inventory, however, suggest that the total number of wells identified in the report (1,338) may be a gross underestimate. This is partially attributable to the reluctance on the part of many well owners to admit to the existence of wells on their property, the ease with which agricultural drainage wells can be constructed, the lack of permit requirements, and the large number of farming operations in the United States (Report to Congress, 1987).

The difficulties and uncertainties inherent in compiling an inventory of agricultural drainage wells lead to several discrepancies in the reported number of wells. For example, in contrast to the 572 agricultural drainage wells reported for the State of Idaho in the 1987 Report to Congress, Graham et al. (1977) claim that over 2000

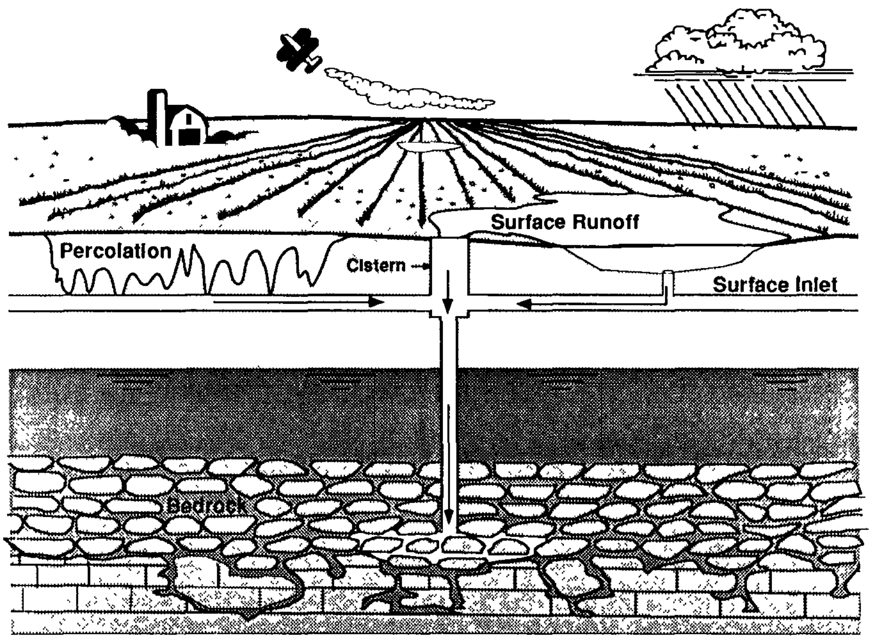

Figure 1. Surface and subsurface runoff entering agricultural drainage well completed into bedrock (modified from Glanville, 1985).

Table 1. States in which presence of agricultural drainage wells has been confirmed (Source: Report to Congress, 1987).

State	EPA Region	No. of Wells Confirmed
New York	II	150
Puerto Rico	II	unknown
West Virginia	III	unknown
Florida	IV	unknown
Georgia	IV	43
Kentucky	IV	unknown
Illinois	V	6
Indiana	V	72
Michigan	V	15
Minnesota	V	54
Oklahoma	VI	unknown
Texas	VI	108
Iowa	VII	230
Missouri	VII	unknown
Nebraska	VII	5
Colorado	VIII	unknown
North Dakota	VIII	1
Idaho	X	572
Oregon	X	16
Washington	X	66

agricultural drainage wells are present within the eastern Snake River plain of Idaho alone. These wells reportedly drain approximately 320,000 acres of land.

In Texas, 300 agricultural drainage well systems are reported to be in operation within a 250 square mile area in southwestern Hidalgo county alone (Texas Water Commission, 1989). This is in contrast to the 108 agricultural drainage wells reported for the State of Texas (1987 Report to Congress). Fluids entering these wells, consisting of irrigation return flow and rainfall runoff, have been observed to contain high concentrations of dissolved solids, nitrate, and the herbicides, bromacil and simazine (Knape, B.K., 1984). Bromacil and simazine, however, have reportedly not impacted local ground water resources (Molofsky, S.J., 1985). An undetermined number of agricultural drainage wells are also known to exist in Runnels County and Oldham County, Texas (Texas Water Commission, 1989).

In contrast to the 230 agricultural drainage wells reported for the State of Iowa in the 1987 Report to Congress, Musterman and Fisher (1981) estimated the existence of between 460 and 920 agricultural drainage wells in northcentral Iowa alone. Glanville (1985) reported that 700 agricultural drainage wells, some more than 100 years old, are estimated to be in use in Iowa. In northcentral Iowa, as many as eight agricultural drainage wells per square mile are reported to exist (Glanville, 1985). Since 1957, permits have been required for the construction of new agricultural drainage wells in Iowa. Only two such permits have reportedly been issued since that date.

The discrepancies and uncertainties noted with regard to the number of agricultural drainage wells in existence appears to reflect the limited information currently available regarding the presence and distribution of agricultural drainage wells. This apparent lack of information stresses the need to carry out more thorough and detailed inventories at the State and Federal levels.

Agricultural drainage well designs

Design of agricultural drainage wells varies from State to State (Report to Congress, 1987). The design depends on whether the wells are to be used for the collection of surface flow, subsurface flow, or both. Figure 2 depicts two typical agricultural drainage well designs. In Iowa, cisterns are generally constructed from poured concrete, grouted bricks, large diameter clay tiles or metal culverts. According to Baker and Austin (1984), most agricultural drainage wells in Iowa either have surface inlets connected to the subsurface drainage systems, or the cisterns are low enough to allow surface drainage to enter the wells directly when pondage occurs.

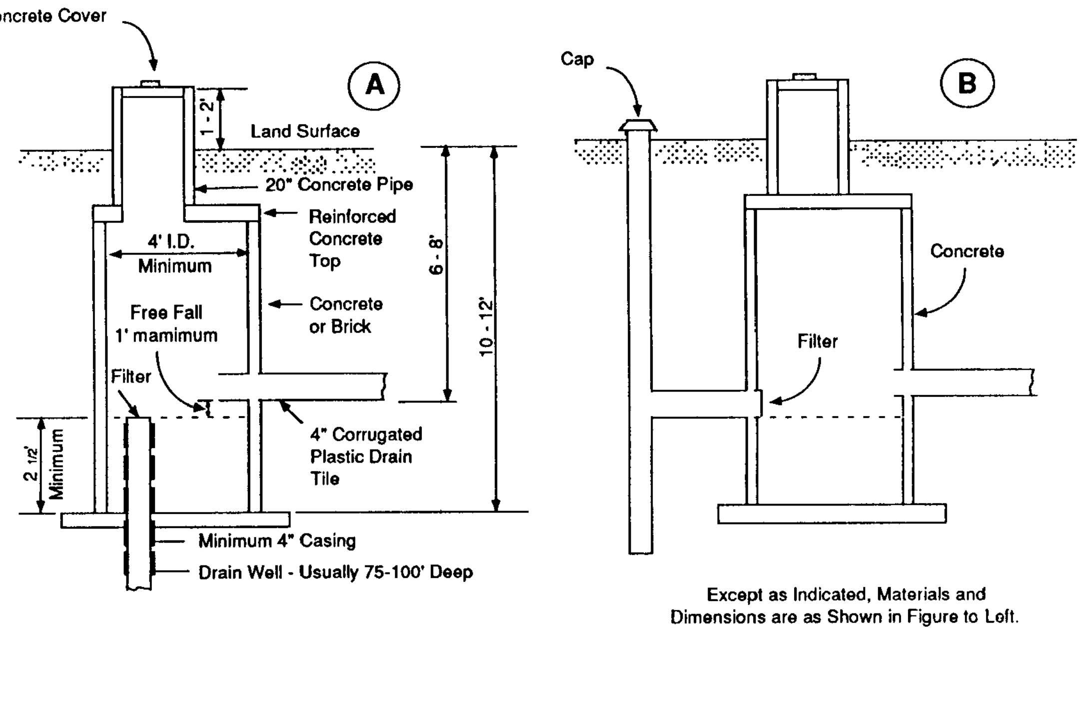

Figure 2. Schematics of agricultural drainage wells (A) with well inside cistern and (B) with well adjacent to cistern (from Texas Department of Water Resources, 1984).

To limit costs, agricultural drainage wells are usually completed in the shallowest permeable zone which can meet the discharge volume requirements. Small-capacity wells may be completed in the vadose zone although most wells are more likely to be completed in shallow bedrock aquifers exhibiting high permeability properties. According to the 1987 Report to Congress, wells in Iowa tend to be completed in fractured, vuggy carbonate formations while wells in Idaho tend to be completed in fractured basalt formations. In efforts to minimize costs, most wells are completed to depths of less than 100 feet. Wells in Idaho are reported to range in depth from 20 to 300 feet with casing depths ranging from 5 to 200 feet below land surface (Report to Congress, 1987). Casing diameters are reported to range from 3 to 24 inches depending on the design capacity of the wells. Large-capacity wells may have screened or inverted inlets, settling ponds, and surface seals. According to Graham et al. (1977), agricultural drainage wells in the eastern Snake River Plain of Idaho are typically 10-30 centimeters (4-12 inches) in diameter, 30-50 meters (100-164 feet) in depth and are capable of accepting flows up to eight cubic meters (282 cubic feet) per minute.

Potential contaminants entering agricultural drainage wells

Potential contaminants entering agricultural drainage wells depend on the particular farming practices in effect and the particular soil type(s) present. Contaminants may include suspended solids, pesticides, fertilizers (e.g. nitrogen and phosphorous compounds), salts, organics, metals, and microbes including pathogens. Surface runoff is generally low in salinity but may contain large quantities of suspended solids, microbes, and pesticides. Contaminants such as pesticides, bacteria, and metals may be attached to suspended solids in surface runoff thereby facilitating the entry of these contaminants into agricultural drainage wells. Subsurface flow is not likely to contain significant levels of suspended solids or bacteria (due to the effects of natural soil filtration processes), but may contain high concentrations of dissolved solids including dissolved pesticides and nutrients (e.g. nitrates). Baker et al. (1985) conducted a study in northcentral Iowa which indicated that nitrate concentrations entering agricultural drainage wells via subsurface flow were as high as 30 mg/l. (The EPA Drinking Water Standard for nitrate as N is 10 mg/l.)

Figure 3 indicates the transport pathways by which contaminants enter agricultural drainage well systems. Table 2 shows ranges for several water quality parameters monitored in runoff entering agricultural drainage wells in northcentral Iowa.

Impact of agricultural drainage wells

The impact of agricultural drainage wells on the subsurface environment will depend on the volume of fluid entering the wells, the type and concentration of contaminants present in the injected fluid, and the nature of the subsurface

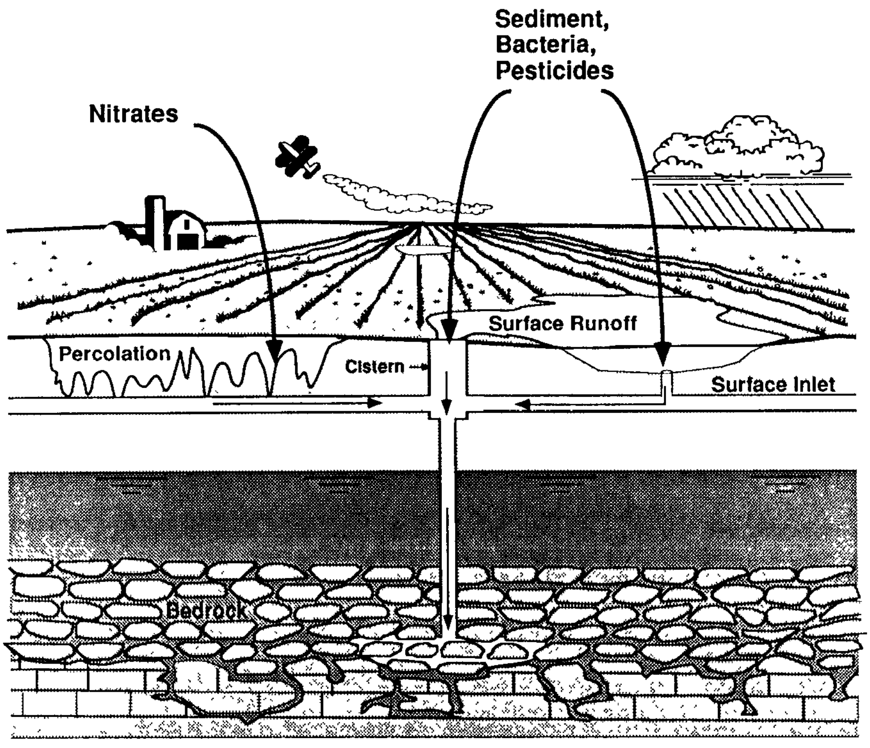

Figure 3. Contaminants and their transport pathways into agricultural drainage wells (modified from Glanville, 1985).

Table 2. Ranges for several water quality parameters monitored in runoff (surface and subsurface) entering four agricultural drainage wells in northcentral Iowa (from Baker et al., 1985).

Constituent (mg/liter)	Range of Concentration
NH_4-N	0.01 - 3.78
NO_3-N	1.50 - 34.0
PO_4-P	0.01 - 1.99
Cl	1.0 - 120.0
Ca	13 - 150
Fe	0.01 - 2.60
Dissolved solids	75 - 745
Suspended solids	0 - 5,360
Atrazine	0 - 0.50
Cyanazine	0 - 80.0
Alachlor	0 - 55.0
Dieldrin	0 - 0.028
Metribuzin	0 - 0.41
Dicamba	0 - 12.0

environment into which the fluid is injected. Since agricultural drainage wells are generally completed in highly permeable formations, such as fractured bedrock, the potential for widespread environmental impact is high. In general, surface runoff can be expected to have the greatest potential negative impact on the subsurface environment both because it may contain high levels of suspended solids, pesticides and bacteria, and because it may be produced in copious volumes. Surface runoff has direct access to the drainage wells and therefore the high permeability formations into which the agricultural drainage wells are completed. Subsurface flow into agricultural drainage wells may exhibit significant impacts in cases where the flow contains high levels of dissolved contaminants (e.g. pesticides, nitrates).

In cases where agricultural drainage wells are completed in the vadose zone, the ability of the injected fluid to potentially impact an underlying aquifer is dependent on the elevation of the injection point relative to the elevation of the water table, the permeability and contaminant attenuation properties of the vadose zone material, the presence of aquicludes and/or aquitards below the area of injection, and the existing quality of the ground water.

For agricultural drainage wells completed in the saturated zone, the likelihood of serious ground-water contamination is considerably more imminent than for drainage wells completed in the vadose zone. Formations deemed attractive for the completion of agricultural drainage wells are often also those formations serving as local supplies of drinking water. The high permeability properties characteristic of these formations can often result in rapid and extensive contamination of downgradient drinking water wells. The extent to which nearby water supply wells may be affected by an agricultural drainage well will depend upon the distance of the injection point from the supply wells, the area of influence of the supply wells, the physical, chemical, and biological contaminant attenuation properties of the formation into which the agricultural drainage wells are completed, the volume of the agricultural drainage fluids being injected, and the types and concentrations of contaminants present in the injected fluid.

Contamination of drinking water supplies arising from the presence of agricultural drainage wells has been reported in Idaho and Iowa. In Iowa, the presence of contaminated supply wells has been reported to coincide with areas highly concentrated with agricultural drainage wells. Baker et al. (1985) conducted a study of the impact of agricultural drainage wells on farm water supply wells in Humboldt and Pocahontas Counties in Iowa. The farm wells were located in three different regions, each about 20 square miles in area. Agricultural drainage wells were prevalent in two of the regions but not in the third. The study showed that one-third of farm wells, in the two regions where large numbers of agricultural drainage wells were present, contained nitrate levels exceeding the recommended drinking water standard. This compared to only 9 percent of farm wells exceeding the nitrate standard in the region not exhibiting a high concentration of agricultural drainage wells. The study also indicated that in most cases, little nitrate contamination was found in water supply wells more than 1.25 miles from an agricultural drainage well. It is assumed that the three different regions studied were similar in geologic character.

In Idaho, water supply wells near agricultural drainage wells have also shown signs of contamination. In a study conducted by Graham (1979), the presence of excessive levels of coliform bacteria in domestic water supplies during the irrigation season was linked to the presence of agricultural drainage wells. Levels of coliform bacteria in excess of Idaho's drinking water standards were observed in 31 percent of the domestic wells in areas exhibiting a high concentration of agricultural drainage wells. The quality of the water was significantly inferior to that of other areas upgradient of the agricultural drainage well areas. In a study conducted by Graham et al. (1977) in the eastern Snake River Plain area of southern Idaho, bacterial levels and turbidity within the recharge zone created by the agricultural drainage wells were far in excess of drinking water standards. Deep percolation of the injected wastewater resulted in bacterial contamination of a deep perched zone and an artesian ground-water system. Suspended solids, as measured by turbidity, were apparently filtered out by the deep percolation process.

Agricultural drainage wells may also be responsible for contamination of supply wells in California. Although agricultural drainage wells have not been inventoried in California, agricultural drainage wells are reported to exist (Report to Congress, 1987; Spencer et al., 1985).

In addition to potentially impacting ground-water resources, agricultural drainage wells may also have an adverse impact on wetlands. Although the impact of agricultural drainage wells on wetlands is beyond the scope of this report, an investigation in this area may be warranted.

Transport and Fate of Agricultural Drainage Well Contaminants

The transport and fate of a particular contaminant entering an agricultural drainage well will depend on the combined properties of the contaminant and the formation into which the contaminant has been introduced. Depending on these properties, the contaminant may be partitioned to various degrees between the different constituents comprising the formation. The partitioning of the contaminant will govern its mobility and therefore its ability to impact the subsurface. Important formation-related properties governing the partitioning process will include permeability, "qualitative" porosity (i.e. presence or absence of extensive void/fracture network), organic content, clay content, microbial activity, redox potential, temperature and pH. These formation properties combined with the specific characteristics of the contaminants will impact on important transport and fate phenomena such as adsorption, filtration and biodegradation.

The contaminants of primary concern in fluids entering agricultural drainage wells include suspended solids, pesticides, microbes, nutrients, salts, and metals. Suspended solids, bacteria, and pesticides are of particular concern in surface runoff while soluble components such as nitrates, salts and some pesticides are of concern in subsurface flow. Insoluble components such as microbes, many pesticides, and suspended solids may, in some cases, conceivably also enter agricultural drainage wells by way of subsurface flow through soil macropores (e.g. channels created by earthworms, root systems, etc.). The size and often extensive network of macropores may occasionally afford such contaminants a relatively unobstructed transport pathway to the subsurface. Macropores provide preferential flow paths and, while normally accounting for only a small percentage of the total soil porosity, may be responsible for a large percentage of the total flow through soils. The significance of macropores in the subsurface transport of contaminants is still under investigation.

The subsurface transport and fate of contaminants known to enter agricultural drainage wells are discussed in the ensuing sections.

A. Suspended Solids

Suspended solids entering agricultural drainage wells are of concern not only because of their objectionable presence in drinking water but because they may also act as a vehicle for the transport of other potential contaminants such as pesticides, bacteria, and metals (i.e. facilitated transport). Pesticides, bacteria, and metals are all capable of strongly sorbing onto the surfaces of suspended solids such as clay minerals and organics. Once in the subsurface, these contaminants may slowly desorb from the suspended solids and contribute to degradation of ground-water quality. The extent to which suspended solids will be transported in the subsurface will depend on the nature of the suspended solids (e.g. particle size, density), the size and pattern of fractures and/or voids in the formation in which the agricultural drainage well is completed, and the local ground-water flow conditions. Suspended solids entering a formation lacking an extensive network of fractures or voids will likely not be transported far and will tend to be filtered out within the immediate vicinity of the well. Lighter, smaller suspended solids entering a formation with an extensive network of fractures and/or voids and a high ground-water flow velocity may conceivably be transported over very large distances. The practice of completing agricultural drainage wells in karst and/or highly fractured bedrock formations which may lack good filtering capabilities, suggests that, in some cases, suspended solids entering agricultural drainage wells may contribute to contamination of underground drinking water sources.

B. Pesticides

Pesticides such as bactericides, fungicides, insecticides, nematocides, rodenticides, and herbicides pose a threat to ground-water supplies because of their toxic properties. Because most pesticides are relatively insoluble and tend to be strongly sorbed to soil particles, pesticides entering agricultural drainage wells are more likely to be associated with surface runoff than subsurface flow. More soluble pesticides such as the anionic herbicides chloramben, 2,4-D and dicamba may, however, be present in subsurface flow. Table 3 provides a list of selected herbicides and their predominant transport modes. Also included in Table 3 is the approximate persistence of each herbicide in soil. Persistence in Table 3 refers to the time required for 90 percent or more of the applied pesticide to disappear from the site of application. The values for persistence provided are, at best, approximations since persistence may be highly variable depending on factors such as climate, soil texture, moisture content, acidity, temperature, and microbiological activity in the soil (EPA-USDA, 1975). Tables A-1, A-2 and A-3 in Appendix A provide a detailed list of pesticides and their predominant transport modes.

According to the 1987 Report to Congress, pesticides commonly detected in significant concentrations in flows entering agricultural drainage wells include atrazine, cyanazine, and metribuzin.

Pesticides exhibiting lower water solubility properties are those more likely to be adsorbed to suspended solid particles. Since suspended solids do enter agricultural drainage wells, they may act as an important vehicle by which pesticides are able to enter the subsurface. The more persistent pesticides such as the chlorinated hydrocarbons, appear to be transported largely attached to sediments (Task Committee on Agricultural Runoff and Drainage of the Water Quality Committee of the Irrigation and Drainage Division, 1977).

Once in the subsurface environment, the transport and fate of a pesticide will depend on the subsurface formation

Table 3. Predominant transport modes and persistence in soils of selected agricultural herbicides (from EPA-USDA Report No. EPA-600/2-75-026a, 1975).

Common Name of Herbicide	Predominant Transport Mode	Approximate Persistence in Soils (days) *
Alachlor	sediment/water	40 - 70
Atrazine	sediment/water	300 - 500
Barban	sediment	20
Bromacil	water	700
Chloramben	water	40 - 60
Cyanazine	sediment/water	--
2,4-D Acid	water	10 - 30
DCPA	sediment	400
Dinoseb	sediment/water	15 - 30
Diuron	sediment	200 - 500
Glyphosate	sediment	150
Metribuzin	water	150 - 200
Simazine	sediment	200 - 400
Trifluralin	sediment	120 - 180

* From reported literature values corresponding to the time required for 90 percent or more of applied pesticide to disappear from sites of application.

properties and the pesticide characteristics. More soluble pesticides not readily susceptible to biodegradation are likely to travel the greatest distances. Pesticide characteristics which will influence the mobility of pesticides in the subsurface include solubility, sorptive characteristics and susceptibility to biodegradation. In general, pesticide biodegradation and adsorption can be expected to decrease with depth due to decreasing microbial activity and decreasing organic content, respectively. Although most pesticides reportedly degrade quite rapidly after application to fields (Ochs, 1980), many pesticides do reach ground water and appear to persist for significant lengths of time. Table 4, which lists pesticides detected in ground water, their ranges, and the number of states in which they have been detected, provides evidence of the wide-scale contamination of ground water by pesticides in the United States. Agricultural drainage wells, in some or many cases, may play a major role.

The transport of pesticides to agricultural drainage wells can be expected to be dependent on a number of factors including the physical and chemical properties of the pesticide, its formulation, the rate and type of application, the crop to which it was applied, tillage practices, topography of the field, weather conditions, and amount and intensity of rainfall following application. Most pesticide residues are found in the top layer of tilled cropland soils, the layer that erodes in the process of sheet erosion (Task Committee on Agricultural Runoff and Drainage of the Water Quality Committee of the Irrigation and Drainage Division, 1977). This suggests that pesticides entering agricultural drainage wells may be correlated with the sediment load.

Spencer et al. (1985) reported on the concentration of pesticides in surface irrigation runoff water following the application of pesticides to large fields of cotton, sugarbeets, alfalfa, lettuce, onions, and canteloupes in the Imperial Valley, California. The concentrations in runoff water were dependent upon the characteristics of the pesticides, their methods and rates of applications, the time elapsed between application and the first irrigation, the number of irrigations since the pesticide application, irrigation efficiency, and other soil management practices. The highest concentrations occurred when herbicides were applied in irrigation water. As would be expected, concentrations of pesticides were also higher in the first irrigation following application. The time elapsed between pesticide application and a given irrigation event was shown to be inversely related to the logarithm of the concentration of the pesticide, suggesting an approximate first-order rate of decrease in runoff concentration with time. Spencer et al. reported that the percentages of the applied pesticides lost in runoff were generally very low, with all seasonal totals for insecticide runoff below 1 percent of the amounts applied, and seasonal losses of soil-applied herbicides usually 1 to 2 percent of the amounts applied. The amount of pesticides in runoff waters did not correlate well with the amounts of sediment in the runoff water, the runoff water volume, or the accumulative water applied.

Spencer et al. also reported that none of the pesticides in the aforementioned study were identified in tile drain effluents at concentrations above minimum detectable levels of 1 to 2 parts per trillion. This suggested that the pesticides used were not sufficiently mobile or persistent to reach the ground water. According to a 1975 U.S. Department of Agriculture (USDA) report, the total amount of pesticide that will generally run off the land during the crop year is less, often much less, than 5 percent of the application. Nevertheless, this small amount of pesticide runoff could still exhibit a significant impact on receiving surface waters and ground water.

Baker et al. (1985) monitored pesticide levels in water entering agricultural drainage wells from row-cropped areas in northcentral Iowa during periods of flow in 1981 and 1982 and observed that pesticide levels were always less than 100 mg/l and usually less than 1 mg/l. The pesticides detected included alachlor, atrazine, carbofuran, chlordane, cyanazine, 2,4-D, dicamba, dieldrin, and metribuzin. These pesticides were detected at maximum concentrations of 55, 0.5, 0.6, 1.8, 80, 0.4, 12, 0.028, and 0.41 μg/liter, respectively. More than half of the samples taken of water draining into the agricultural drainage wells did not show detectable levels of pesticides. It is not known in the study cited whether pesticides associated with suspended solids were included in the analyses.

Pesticides may indirectly add other pollutants to soil and water (U.S. Department of the Interior, 1969). Organic

Table 4. Confirmed presence of pesticides in ground water of 17 different States (from Hallberg, 1987; after Cohen et al., 1986).

Pesticide Name	Typical Concentration (mg/l)	MCL (m) or Proposed MCL (p) (mg/l)	Number of States
Herbicides			
alachlor	0.10 - 10.0	2 (p)	4
atrazine	0.3 - 3.0	3 (p)	5
bromacil	300		1
cyanazine	0.1 - 1.0		2
DCPA (and acid products)	50.0 - 700.0		
dinoseb	1.0 - 5.0	7 (m)	1
metolachlor	0.1 - 0.4		2
metribuzin	1.0 - 4.0		1
simazine	0.2 - 3.0	1 (m)	3
Insecticides and Nematicides			
aldicarb (sulfoxide and sulfone)	1.0 - 50.0	10 (p)	15
carbofuran	1.0 - 50.0	40 (p)	3
DBCP	0.2 - 20.0	0.2 (p)	5
1,2-DCP	1.0 - 50.0	5 (p)	4
dyfonate	0.1		1
EDB	0.05 - 20.0	0.05 (p)	8
oxamyl	5.0 - 65.0	200 (m)	2
1,2,3-trichloropropane (impurity)	0.1 - 5.0		2

pesticides can contain metals such as mercury, zinc, manganese, copper, chromium, cadmium and tin which can be released during decomposition.

C. Nutrients

Nutrients, in the form of nitrogen and phosphorous compounds, are essential to plant growth and are commonly applied in agricultural regions as fertilizer. Nitrogen, in the form of nitrate-nitrites, is of greatest concern in drinking water because of its often high concentrations and its link to methemoglobinemia in infants. High concentrations of nitrate may also have adverse effects on livestock and may stimulate eutrophic processes in surface waters. Nitrates-nitrites are highly soluble and are not readily adsorbed by soils. They can therefore be expected to be present in both surface and subsurface flows entering agricultural drainage wells. Often nitrates-nitrites are added directly to irrigation water.

A previously cited study conducted by Baker et al. (1985) in northcentral Iowa indicated that farm wells within 2 km of drainage wells showed elevated levels of nitrates. Some elevated levels were observed in areas with an unsaturated zone thickness of 15 meters or more. Other areas, where the unsaturated zone thickness was 15 meters or more and where there were no drainage wells nearby, reportedly did not show elevated levels of nitrates. These observations were cited as evidence that the presence of nitrates, in this study, were likely attributable to agricultural drainage wells rather than the infiltration of nitrates from the land surface.

As part of their study, Baker et al. also observed that during periods between runoff events, when all the drainage to the agricultural drainage wells was subsurface drain flow, nitrate concentrations were highest in the range 10-30 mg/l. When the agricultural drainage wells received both surface and subsurface runoff during periods of snowmelt and rainfall, concentrations of nitrate often dropped below 10 mg/l. Concentrations of nitrates entering the agricultural drainage wells were observed to exceed the Federal Primary Drinking Water Standard of 10 mg/l for 85 percent of the samples with an overall average of 16 mg/l.

A compilation of data from the Big Spring basin aquifer in northeastern Iowa indicated that in the 1930s, nitrate concentrations in the aquifer were less than 1 mg/l. In the 1950s and 1960s, the nitrate concentrations in the aquifer averaged about 3 mg/l and by 1983, the average concentration was 10.1 mg/l. The increases in nitrate concentrations were reported to directly parallel increases in the amount of nitrogen fertilizer applied (Hallberg, 1987). Although the extent to which agricultural drainage wells (if present in northeastern Iowa) may have contributed to the observed increases in nitrate concentrations is not known, it can be assumed that the presence of agricultural drainage wells may have facilitated the increased contamination trend.

Phosphorus in fertilizers does not appear to present a significant concern with regard to contamination of ground water by agricultural drainage wells. Phosphorous is not toxic to man or animals in the forms commonly found in water. Phosphorous (in the form of phosphate) is readily adsorbed by inorganic and organic soil matter. Surface runoff containing high levels of suspended solids may, however, exhibit high concentrations of adsorbed phosphorous. If drained into highly fractured or karst formations, the phosporous could conceivably reach surface waters and stimulate eutrophic processes.

D. Microbes

The introduction of microbes (e.g. bacteria, viruses) into ground waters is of concern where surface runoff enters agricultural drainage wells. Microbial populations entering agricultural drainage wells may include pathogenic organisms which cause diseases such as bacillary and amoebic dysentry, cholera, typhoid and paratyphoid fever, bacterial gastroenteritis, infectious hepatitis, and poliomyelitis. Graham et al. (1977), identified sediment loads and bacteria from return flows as the most serious threat to ground water quality in a study conducted in Idaho. Table 5 indicates the levels of bacteria detected in surface return flow in the study. The levels of bacteria detected were very high relative to the EPA Drinking Water Standards.

The migration of bacteria in the subsurface to potential supply wells will be governed by the subsurface formation characteristics. In the presence of extensive formation voids and/or fractures, bacteria may travel significant distances. In a previously cited study conducted in southeast Minidoka County, Idaho, Graham et al. (1979) reported that excessive levels of indicator bacteria were observed in domestic ground-water supplies and were apparently attributable to the discharge of irrigation wastewater to nearby agricultural drainage wells. Total coliform and fecal coliform counts with highs of 284 and 27, respectively, were detected in domestic water supply wells. In at least one study area, the detection of indicator bacteria coincided closely with discharges to agricultural drainage wells located one half mile or more away. The apparent significant migration of bacteria observed is likely attributable to the fractured nature of the basalt formations into which agricultural drainage wells in southeast Minidoka County are completed.

E. Salts

The addition of fertilizers and soil conditioners to crops may contribute substantial amounts of the major ions to runoff (Seitz et al., 1977). Major ions may include calcium, magnesium, sodium, potassium, chloride, sulfate, and carbonate. Some soils, in addition, may already exhibit relatively high natural salt contents. Evaporation, transpiration, and recycling of irrigation waters tend to concentrate the major ions such that the levels entering agricultural drainage wells may be of concern. Sulfate concentrations in excess of 250 mg/l, for example, may be cathartic (i.e. exhibit a laxative effect). Salts are more likely to enter agricultural drainage wells via subsurface flow.

Table 5. Levels of bacteria detected in surface return flow (from Report to Congress, 1987; after Graham et al., 1977). The EPA Drinking Water Standard for total coliforms is one organism/100 ml.

Parameter	Number of Determinations	Low	Mean	High
Total coliforms (organisms/100 ml)	45	580	29,000	96,000
Fecal coliforms (organisms/100 ml)	45	65	850	13,000
Fecal streptococci (organisms/100 ml)	38	900	7,400	16,000

F. "Incidental" contaminants

A potential concern, where applicable, is the presence of agricultural drainage wells near roadsides, equipment preparation or maintenance areas, or other trafficked areas susceptible to accidental or intentional spillage/discharge of environmentally objectionable materials. The presence of agricultural drainage wells in these areas would provide virtually unobstructed conduits for the movement of potentially hazardous materials to the subsurface. The extent to which such subsurface contamination scenarios may occur is not known.

Discussion

The evidence that agricultural practices can lead to the contamination of ground water is undeniable. The use of pesticides and fertilizers has increased dramatically over the past 20 to 30 years. Agricultural pesticide use in the United States, for example, has more than doubled since 1964. Nitrogen fertilizer use on corn has increased from 72 kg/ha in 1965 to over 150 kg/ha in 1982 (Hallberg, 1987). Contamination of the subsurface from agricultural practices is being increasingly documented. Examples include California where 3,000 supply wells have been observed to be contaminated with 57 pesticides of which 22 have been traced to agricultural use (Ground Water Monitor, 1985). In Iowa, where agricultural drainage wells are commonly used, pesticides are more commonly found in ground water than are industrial chemicals (Ground Water Monitor, 1985). The increasing number of agriculture-related ground-water contamination scenarios being documented warrants increasing concern.

The specific contribution of agricultural drainage wells to the increasing number of ground-water contamination scenarios being documented nation-wide is presently not known. However, it must be assumed that the presence of agricultural drainage wells can only serve to facilitate subsurface contamination. Confirmed cases of ground-water contamination by agricultural drainage wells have been identified in Iowa and Idaho based on studies cited herein. This confirmed evidence suggests that either alternatives to the use of agricultural drainage wells should be sought or the quality of fluids entering agricultural drainage wells should be better controlled.

Possible solutions to the problems associated with the use of agricultural drainage wells are as follows:

1. The complete elimination of agricultural drainage wells and a return to natural drainage conditions.

2. The complete elimination of agricultural drainage wells with drainage of excess water being directed to surface waters.

3. The elimination of surface water inlets associated with agricultural drainage well systems.

4. Control of surface runoff and surface erosion.

5. Improved fertilizer and pesticide management.

6. Pretreatment of surface runoff followed by recycling and/or controlled disposal to agricultural drainage wells.

The complete elimination of agricultural drainage wells and a return to natural drainage conditions would likely reduce the impact of agricultural drainage on ground water in most hydrogeologic regimes. However, elimination of wells would potentially result in serious socio-economic repercussions. Kanwar et al. (1986) predict that the elimination of agricultural drainage wells in northcentral Iowa would cost the farmers of the area $270 per hectare per year in crop production. Glanville (1985) predicts that the elimination of agricultural drainage wells in northcentral Iowa would result in a drop in corn yields by an average 50 bushels per acre and a drop in soybeans by an average 18 bushels per acre. Additional economic burdens according to Glanville would include mired farm equipment, delayed planting and harvesting, and fertilizer losses.

The construction of alternative drainage systems allowing for the drainage of excess water to surface waters has been suggested as a possible solution (Kanwar et al., 1986). Glanville (1985) claims this could be expensive in some cases with capital costs in northcentral Iowa ranging from $100 per acre on land lying near existing drainage facilities to over $300 per acre on remote fields requiring pumped drainage or deep ditches and tile mains. According to Kanwar et al. (1986), the costs in northcentral Iowa would range from $200 to $964 per hectare. However, even if the implementation of such alternative drainage systems were economically feasible, the potential impact of the drainage water on receiving surface waters must be considered. Nutrients entering surface waters, for example, may give rise to serious eutrophication problems while pesticides may exhibit toxic effects to aquatic plant and animal life.

Options for the control of surface runoff and surface erosion could include discouraging the practice of applying excess quantities of irrigation water. Measures which may reduce the volumes of surface runoff generated in drier regions include irrigation scheduling, the use of high-efficiency irrigation methods, alternate furrow irrigation, and the use of drought tolerant crops (if possible). In those regions of the nation where "water appropriation rights" are in effect and excess irrigation water is applied in order to avoid loss or reduction of future water allocations, efforts should be made to change the governing rules or regulations to discourage such practices. Table 6 provides a list of additional options which could be considered in controlling surface runoff and erosion. Table 7 provides some comments regarding the practical application of the various options. For detailed

Table 6. Practices for controlling direct runoff and their effectiveness (from EPA-USDA Report No. EPA-600/2-75-026a, 1975)

Runoff Control Practice	Effectiveness of Practice
No-till plant in prior crop residues	Variable effect on direct runoff from substantial reductions to increases on soils subject to compaction.
Conservation tillage	Slight to substantial runoff reduction.
Sod-based rotations	Substantial runoff reduction in sod year; slight moderate reduction in rowcrop year.
Meadowless rotations	None to slight runoff reduction.
Winter cover crop	Slight runoff increase to moderate reduction.
Improved soil fertility	Slight to substantial runoff reduction depending on existing fertility level.
Timing of field operations	Slight runoff reduction.
Plow plant systems	Moderate runoff reduction.
Contouring	Slight to moderate runoff reduction.
Graded rows	Slight to moderate runoff reduction.
Contour strip cropping	Moderate to substantial runoff reduction.
Terraces	Slight increase to substantial runoff reduction.
Grassed outlets	Slight runoff reduction.
Ridge planting	Slight to substantial runoff reduction.
Contour listing	Moderate to substantial runoff reduction.
Change in land use	Moderate to substantial runoff reduction.
Other practices	
Contour furrows	Moderate to substantial reduction.
Diversions	No runoff reduction.
Drainage	Increase to substantial decrease in surface reduction.
Landforming	Increase to slight runoff reduction.
Construction of ponds	None to substantial runoff reduction. Relatively expensive. Good pond sites must be available. May be considered as a treatment device.

discussions of the options listed, the reader is referred to the joint EPA-USDA publication "Control of Water Pollution from Cropland; Volume 1" (1975).

The elimination of surface water inlets on agricultural drainage wells to prevent direct entry of surface water runoff would be expected to significantly reduce the impact of agricultural drainage wells on ground-water quality. Surface runoff generally carries the bulk of contamination (in the form of suspended solids, bacteria and pesticides) and is therefore of greatest concern. The elimination of surface water inlets may, however, give rise to surface ponding problems and thereby significantly retard land drainage. Crop yields would likely be impacted, although this impact would probably be less than the impact which would result from total elimination of agricultural drainage wells.

Improved fertilizer and pesticide management would serve to reduce the amount of nutrients and pesticides entering agricultural drainage wells. In Iowa, it has been suggested that fall application of nitrogen be discouraged since significant amounts are leached out of the root zone by spring rains before crops can use it. Spring and early

Table 7. Principal types of cropland erosion control practices and their effectiveness (from EPA-USDA Report No. EPA-600/2-75-026a, 1975).

Erosion Control Practice	Effectiveness of Practice
No-till plant in prior-crop residues	Most effective in dormant grass or small grain; highly effective in crop residues; minimizes spring sediment surges and provides year-round control; reduces man, machine and fuel requirements; delays soil warming and drying; requires more pesticides and nitrogen; limits fertilizer- and pesticide-placement options; some climatic and soil restrictions.
Conservation tillage	Includes a variety of no-plow systems that retain some of the residues on the surface.
Sod-based rotations	Good meadows lose virtually no soil and reduce erosion from succeeding crops; total soil loss greatly reduced but losses unequally distributed over rotation cycle; aid in control of some diseases and pests; more fertilizer-placement options; less realized income from hay years; greater potential transport of water soluble P; some climatic restrictions.
Meadowless rotations	Aid in disease and pest control; may provide more continuous soil protection than one-crop systems.
Winter cover crops	Reduce winter erosion where corn stover has been removed and after low-residue crops; provide good base for slot-planting next crop; usually no advantage over heavy cover of chopped stalks or straw; may reduce leaching of nitrate; water use by winter cover may reduce yield of cash crop.
Improved soil fertility	Can substantially reduce erosion hazards as well as increase crop yields.
Timing of field operations	Fall plowing facilitates more timely planting in wet springs, but it greatly increases winter and early spring erosion hazards; optimum timing of spring operations can reduce erosion and increase yields.
Plow-plant systems	Rough, cloddy surface increases infiltration and reduces erosion; seedling stands may be poor when moisture conditions are less than optimum. Mulch effect is lost by plowing.
Contouring	Can reduce average soil loss by 50% on moderate slopes, but less on steep slopes; loses effectiveness if rows break over; must be supported by terraces on long slopes; soil, climatic, and topographic limitations; not compatible with use of large farming equipment on many topographies. Does not affect fertilizer and pesticide rates.
Graded rows	Similar to contouring but less susceptible to row breakovers.
Contour strip cropping	Rowcrop and hay in alternate 50- to l00-foot strips reduce soil loss to about 50% of that with the same rotation contoured only; fall seeded grain in lieu of meadow about half as effective; alternat ing corn and spring grain not effective; area must be suitable for across-slope farming and establishment of rotation meadows.
Terraces	Support contouring and agronomic practices by reducing effective slope length and runoff concentration; reduce erosion and conserve soil moisture; facilitate more intensive cropping: conventional gradient terraces often incompatible with use of large equipment, but new designs have alleviated this problem; substantial initial cost and some maintenance costs.
Grassed outlets	Facilitate drainage of graded rows and terrace channels with minimal erosion; involve establishment and maintenance costs and may interfere with use of large implements.
Ridge planting	Earlier warming and drying of row zone; reduces erosion by concentrating runoff flow in mulch-covered furrows; most effective when rows are across slope.
Contour listing	Minimizes row breakover; can reduce annual soil loss by 50%; loses effectiveness with post-emergence corn cultivation.
Change in land use	Sometimes the only solution. Well managed permanent grass or woodland effective where other control practices are inadequate: lost acreage can be compensated for by more intensive use of loss erodible land.
Other practices	Contour furrows, diversions, subsurface drainage, land forming, closer row spacing, etc.

summer nitrogen applications would improve nitrogen utilization. It has also been suggested that three well-timed nitrogen applications rather than a single pre-plant application be considered. This would result in a 35 percent reduction in nitrate levels (Glanville, 1985). Reductions in the quantities of fertilizers and pesticides applied would also significantly reduce impacts on ground water. According to Hallberg (1986), a reduction in N-fertilizer application rates of 90 kg-N/ha in Hall County, Nebraska, resulted in no reduction in corn yields over four years. Hallberg also reports that recent studies in Pennsylvania show that some current methods recommend fertilizer-N applications more than 100 kg-N/ha greater than rates that would produce economic optimum production. In contrast, however, according to figures provided by Baker and Austin in 1984, decreasing the nitrogen application rate in Iowa from 150 to 75 kg/ha was predicted to decrease net return for corn by about $26/acre. Other measures which may reduce the quantity of fertilizer and pesticide contaminants entering agricultural drainage wells might include the use of slow-release fertilizers and the development and use of more biodegradable pesticides. Tables 8 and 9 provide options which may be considered for controlling pesticide and fertilizer loss. For detailed discussions on the various

Table 8. Practices for the control of pesticide loss from agricultural applications and their effectiveness (from EPA-USDA Report No. EPA-600/2-75-026a, 1975)

Pesticide Control Practice	Effectiveness of Practice
Broadly Applicable Practices	
Using alternative pesticides	Applicable to all field crops; can lower aquatic residue levels; can hinder development of target species resistance.
Optimizing pesticide placement with respect to loss	Applicable where effectiveness is maintained; may involve moderate cost.
Using crop rotation is planted.	Universally applicable; can reduce pesticide loss significantly; some indirect cost if less profitable crop
Using resistant crop varieties	Applicable to a number of crops; can sometimes eliminate need for insecticide and fungicide use; only slight usefulness for weed control.
Optimizing crop planting time	Applicable to many crops; can reduce need for pesticides; moderate cost possibly involved.
Optimizing pesticide formulation	Some commercially available alternatives; can reduce necessary rates of pesticide application.
Using mechanical control methods	Applicable to weed control; will reduce need for chemicals substantially; not economically favorable.
Reducing excessive treatment	Applicable to insect control; refined predictive techniques required.
Optimizing time of day for pesticide application	Universally applicable can reduce necessary rates of pesticide application.
Practices Having Limited Applicability	
Optimizing date of pesticide application	Applicable only when pest control is not adversely affected; little or no cost involved.
Using integrated control programs	Effective pest control with reduction in amount of pesticide used; program development difficult.
Using biological control methods	Very successful in a few cases; can reduce insecticide and herbicide use appreciably.
Using lower pesticide application rates	Can be used only where authorized; some monetary savings.
Managing aerial applications	Can reduce contamination of non-target areas.
Planting between rows in minimum tillage	Applicable only to row crops in non-plow based tillage; may reduce amounts of pesticides ncecssary.

Table 9. Practices for the control of nutrient loss from agricultural applications and their effectiveness (from EPA-USDA Report No. EPA-600/2-75-026a, 1975)

Nutrient Control Practice	Effectiveness of Practice
Eliminating excessive fertilization	May cut nitrate leaching appreciably, reduces fertilizer costs; has no effect on yield.
<u>Leaching Control</u>	
Timing nitrogen application	Reduces nitrate leaching; increases nitrogen use efficiency; ideal timing may be less convenient.
Using crop rotations	Substantially reduces nutrient inputs; not compatible with many farm enterprises; reduces erosion and pesticide use.
Using animal wastes for fertilizer	Economic gain for some farm enterprises; slow release of nutrients; spreading problems.
Plowing-under green legume crops	Reduces use of nitrogen fertilizer; not always feasible.
Using winter cover crops	Uses nitrate and reduces percolation; not applicable in some regions; reduces winter erosion.
Controlling fertilizer release of transformation	May decrease nitrate leaching; usually not economically feasible; needs additional research and development.
<u>Control of Nutrients in Runoff</u>	
Incorporating surface applications	Decreases nutrients in runoff; no yield effects; not always possible; adds costs in some cases.
Controlling surface applications	Useful when incorporation is not feasible.
Using legumes in haylands and pastures	Replaces nitrogen fertilizer; limited applicability; difficult to manage.
<u>Control of Nutrient Loss by Erosion</u>	
Timing fertilizer plow-down	Reduces erosion and nutrient loss; may be less convenient.

options listed, the reader is again referred to the joint EPA-USDA publication "Control of Water Pollution from Cropland; Volume 1" (1975).

Pretreatment of surface runoff and subsequent recycling and/or controlled discharge of treatment effluents may also significantly reduce the level of contaminants entering agricultural drainage wells. Surface runoff, because of its often high suspended solids, pesticide and bacteria content, is likely to exhibit a greater impact on ground water than subsurface flow. A pretreatment process might simply involve the use of a constructed settling pond which would subject the surface runoff entering the pond to a certain residence time. Settling of suspended solids could be induced and the supernatant could then be allowed to enter the agricultural drainage well. Alternatively, where feasible, the supernatant could be recycled. Issues of concern pertaining to the implementation of pretreatment settling ponds include potential difficulties in siting ponds and potentially high construction costs.

Recommendations

In addressing the problem of contamination of ground water by agricultural drainage wells, some States have proceeded to recommend guidelines for protecting USDW in areas near agricultural drainage wells (Report to Congress, 1987). These guidelines include:

1. Locating and proper plugging of all abandoned wells within the immediate area of agricultural drainage wells (Iowa);

2. Requiring that fluids meet drinking water standards at the point of injection (Nebraska, Oregon);

3. Requiring irrigation tailwater recovery and pumpback (Oregon);

4. Reducing the volume of irrigation return flow (where applicable) by applying only the quantity of water necessary (California);

5. Closing surface inlets in order to allow infiltration through soil to decrease the transport of bacteria, some pesticides, and sediment to the aquifer (Missouri);

6. Raising the inlets above the maximum ponding levels (Iowa);

7. Discouraging use and encouraging elimination of agricultural drainage wells by developing alternative drainage methods (Iowa).

Further efforts should be made at the State and Federal levels to generate a more accurate inventory of agricultural drainage wells currently in existence. Once a more accurate inventory can be obtained, rational decisions regarding potential modifications or alternatives to agricultural drainage wells can be made. A more accurate inventory will also provide an opportunity to better correlate the presence of agricultural drainage wells with specific ground-water contamination scenarios.

The practice of applying excess fertilizer and pesticide should be curtailed as a first step in addressing not only the problem of agricultural contaminants entering agricultural drainage wells, but in addressing the problem of ground-water contamination due to agricultural practices, in general. Current federal programs tolerate and sometimes encourage inefficient fertilizer and pesticide use. Government grading standards require fruits and vegetables to meet stringent cosmetic standards with little bearing on nutritional quality yet at the expense of adding excess pesticides (Ground Water Monitor, Sept. 12, 1989). Alternative agricultural practices designed to reduce fertilizer and pesticide use may include diversification rather than continuous planting of fields to a single crop or few crops, biological pest control, and genetic improvements in crops to resist pests and disease.

Studies should probably be conducted to determine the feasibility of using pretreatment settling ponds in treating surface runoff. Studies should include determination of the ease of pond siting and construction, average costs of pond construction, sediment removal effectiveness, and quality of supernatant generated for recycling and/or disposal. A pilot study at a selected location in Iowa or Idaho may be warranted.

Studies should probably also be conducted to establish the feasibility of achieving more optimal designs for agricultural drainage well systems. These studies should focus on systems designed to improve injectate quality. More optimal designs might include systems which will maximize the subsurface flow component and minimize surface runoff while still providing adequate drainage.

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APPENDIX A

Table A-1. Agricultural herbicides: Types, transport modes, toxicities, and persistence in soil (from EPA-USDA Report No. EPA-600/2-75-026a, 1975)

Common Names of Herbicides	Chemical Class [1]	Predominant Transport Mode	Toxicity [3]		Approximate Persistence in Soil, days
			Rat, Acute Oral LD_{50} mg/kg	Fish[4] LC_{50}, mg/liter	
Alachlor	AM	SW	1200	2.3	40-70
Ametryne[5]	TZ	SW	1110	Low toxicity	30-90
Amitrole	TZ	W	2500	>50	15-30
Asulam	CB	W	>8000	[6]5000	25-40
Atrazine	TZ	SW	3080	12.6	300-500
Barban	CB	S	1350	[7]1.3	20
Benefin	NA	S	800	[6]0.03	120-150
Bensulide	AM	S	770	0.72	500-700
Bentazon	DZ	W	1100	190	
Bifenox	AR	S	4600	1.8	40-60
Bromacil	DZ	W	5200	70	700
Bromoxynil	NT	SW	250	0.05	
Butylate	CB	S	4500	4.2	40-80
Cacodylic Acid	AS	S		[8]>40	
CDAA	AM	W	700	2.0	20-40
CDEC	CB	SW	850	4.9	20-40
Chloramben	AR	W	3500	[6]7.0	40-60
Chlorbromuron	UR	SW	2150	0.56	
Chloroxuron	UR	S	3700	[8]>15	300-400
Chlorpropham	CB	SW	1500	[6]10	120-260
Cyanazine	TZ	SW	334	4.9	
Cycloate[5]	CB	SW	2000	4.5	120-220
2, 4-D Acid	PO	W	370	[9]>50	10-30
2, 4-D Amine	PO	W	370	[8]>15	10-30
2, 4-D Ester	PO	S	500-875	[8]4.5	10-30
Dalapon	AL	W	6590	>100	15-30
2,4-DB	PO	S	300	4	
DCPA	AR	S	3000	>500	400
Diallate	CB	S	395	5.9	120
Dicamba	AR	W	1028	35	
Dichlobenil	NT	S	3160	10-20	60-180
Dinitramine	NA	S	3000	6.7	90-120
Dinoseb	PH	SW	5	[7,10]0.4	15-30
Diphenamid	AM	W	970	25	90-180
Diquat	CT	S	400	12.3	>500
Diuron	UR	S	3400	>60	200-500
DSMA	AS	S	600	>15	
Endothall	PH	W	38	1.15	
EPTC	CB	SW	1360	19	30
Fenac[5]	AR	SW	1780	7.5	350-700
Fenuron	UR	W	6400	53	30-270
Fluometuron	UR	SW	7900	[10]>60	
Fluorodifen	AR	S	15000	0.18	
Glyphosate	AL	S	4320	Low toxicity	150
Isopropalin	NA	S	5000	Toxic	150
Linuron	UR	S	1500	16	120
MBR 8251	AM	SW	633	312	
MCPA	PO	SW	650	10	30-180
Metribuzin	TZ	W	1930	>100	150-200
Molinate	CB	W	501	0.29	80
Monuron	UR	SW	3500	1.8	150-350
MSMA	AS	S	700	>15	
Naptalam	AR	W	1770	>180	20-60

Table A-1. Agricultural herbicides: Types, transport modes, toxicities, and persistence in soil (continued)

Common Names of Herbicides	Chemical Class[1]	Predominant Transport Mode[2]	Toxicity[3]: Rat, Acute Oral LD_{50}, mg/kg	Toxicity[3]: Fish[4] LC_{50}, mg/liter	Approximate Persistence in Soil, days
Nitralin	NA	S	2000	Low toxicity	
Nitrofen	PO	S	2630	Toxic	
Oryzalin	AM	S	>10000	Low toxicity	
Paraquat	CT	S	150	[6]400	>500
Pebulate[5]	CB	S	921	[11]6.3	50-60
Phenmedipham	CB	S	2000	[10]20	100
Picloram	AR	W	8200	2.5	550
Profuralin	NA	S	2200	Toxic	320-640
Prometone[5]	TZ	S	1750	[9]>1	>400
Prometryne[5]	TZ	S	3750	[9]>1	30-90
Pronamide[5]	AM	S	5620		60-270
Propachlor[5]	AM	W	710	1.3	30-50
Propanil[5]	AM	S	1384	>10	1-3
Propazine[5]	TZ	S	5000	>100	200-400
Propham	CB	W	5000	[6]32	20-60
Pyrazon	DZ	W	2500	[12]40	30-60
Silvex	PO	SW	375	[9]0.36	
Simazine	TZ	S	5000	5	2000-400
2, 4, 5 -T	PO	W	300	0.5-16.7	
TCA	AL	W	3370	[13]>2000	20-70
Terbacil	DZ	W	5000	[14]86	700
Terbutryne[5]	TZ	SW	2400	Low toxicity	20-70
Triallate[5]	CB	S	1675	4.9	30-40
Trifluralin	NA	S	3700	[6]0.1	120-180
Vernolate[5]	CB	SW	1625	9.6	50

[1] Chemical type designations: AL, aliphatic acids; AM, amides and anilides; AR, aromatic acids and esters; AS, arsenicals; CB, carbamates and thiocarbamates; CT, cationics; DZ, diazines; NA, nitroanilines; NT, nitriles; PH, phenols and dicarboxylic acids; PO, phenoxy compounds; TZ, triazines and triazoles; UR, ureas.

[2] Where movement of herbicides in runoff from treated fields occurs, S denotes those chemicals that will most likely move primarily with the sediment, W denotes those that will most likely move primarily with the water, and SW denotes those that will most likely move in appreciable proportion with both sediment and water.

[3] Expressed as the lethal dose, or lethal concentration, to 50% of the test animals (LD_{50} or LC_{50}, respectively).

[4] 48- or 96-hour LC_{50} for bluegills or rainbow trout, unless otherwise specified.

[5] Trade name; no corresponding common name exists.

[6] 24-hour LC_{50}

[7] For goldfish.

[8] For killifish.

[9] For spot.

[10] LC_{100}.

[11] For mullet

[12] For harlequin fish.

[13] For catfish.

[14] For sunfish.

Table A-2. Agricultural insecticides and miticides: Types, transport modes, and toxicities (from EPA-USDA Report No. EPA-600/2-75-026a, 1975)

Common Names of Insecticides-Miticides	Chemical Class	Predominant Transport Mode	Toxicity[3] Rat, Acute Oral LD_{50} mg/kg	Toxicity[3] Fish[4] LC_{50} mg/liter
Aldicarb[5]	CB	W	0.93	
Aldrin	OCL	S	35	0.003
Allethrin	PY	S	680	0.019
Azinphos ethyl[6]	OP	S	7	0.019
Azinphos methyl	OP	S	11	0.01
Benzene hexachloride	OCL	S	1000	0.79
Binapacryl	N	U	120	0.04
Bux[6]	CB	S	87	0.29
Carbaryl	CB	SW	500	1
Carbofuran[5]	CB	W	8	0.21
Carbophenothion	OP	S	10	0.23
Chlorbenside	S	S	3000	
Chlordane	OCL	S	335	0.01
Chlordimeform	N	W	162	1
Chlorobenzilate[6]	OCL	S	700	0.71
Chlorpyrifos	OP	U	97	0.02
DDT	OCL	S	113	0.002
Demeton[5]	OP	W	2	0.081
Diazinon[5,6]	OP	SW	76	0.03
Dicofol[6]	OCL	S	684	0.1
Dicrotophos	OP	W	22	8
Dieldrin	OCL	S	46	0 003
Dimethoate	OP	W	185	9 .6
Dioxathion	OP	S	23	0.014
Disulfoton	OP	S	2	0 04
Endosulfan	OCL	S	18	0 001
Endrin	OCL	S	7.3	0.0002
EPN	OP	S	8	0.1
Ethion	OP	S	27	0.23
Ethoprop	OP	U	61.5	1
Fensulfothion[5]	OP	SW	2	[7]0.15
Fonofos[6]	OP	S	8	0.03
Heptachlor	OCL	S	90	0 009
Landrin[6]	CB	SW	178	0.95
Lindane	OCL	S	88	0 018
Malathion	OP	W	480	0.019
Metaldehyde	O	W	1000	> 100.0
Methidathion	OP	U	25	
Methomyl	CB	U	17	~0 9
Methoxychlor	OCL	S	5000	0.007
Methyl demeton[6]	OP	W	65	4
Methyl parathion[6]	OP	SW	9	1.9
Mevinphos	OP	W	4	0 017
Mexacarbate	CB	SW	22.5	1 73
Monocrotophos	OP	W	21	7
Naled	OP	S	250	0 078
Ovex	S	S	2000	0 7
Oxythioquinox	S	S	1100	0.096
Parathion	OP	S	4	0.047
Perthane[6]	OCL	S	>4000	0 007
Phorate[5]	OP	SW	1	0 0055
Phosalone	OP	S	96	3 4
Phosmet[6]	OP	S	147	0.03

Table A-2. Agricultural insecticides and miticides: Types, transport modes, and toxicities (continued).

Comon Names of Insecticides-Miticides	Chemical Class[1]	Predominant Transport Mode[2]	Toxicity[3] Rat Acute Oral LD_{50}, mg/kg	Fish4 LC_{50}, mg/liter
Phosphamidon	OP	W	11	8
Propargite[6]	S	U	2200	0.03
Propoxur	CB	W	95	[9]0.025
TDE	OCL	S	3360	0.009
TEPP	OP	W	I	[9]0.39
Tetrachlorvinphos	OP	S	4000	0.53
Tetradifon	OCL	SW	14000	1.1
Thionazin	OP	W	12	[7]0.10
Toxaphene	OCL	S	69	0.003
Trichlorfon	OP	W	275	.0.16

1 Chemical type designations: CB. carbamates; N, miscellaneous nitrogenous compounds; O, cyclic oxygen compounds; OCL,organochlorines; OP, organophosphorus compounds; PY, synthetic pyrethrin; S, aromatic and cyclic sulfur compounds.

2 Where movement of insecticides in runoff from treated fields occurs, S denotes those chemicals that will most likely move primarily with the sediment, W denotes those that will most likely move primarily with the water, SW denotes those that will most likely move in appreciable proportion with both sediment and water, and U denotes those whose predominant mode of transport cannot be predicted because properties are unknown.

3 Expressed as the lethal dose, or lethal concentration, to 50% of the test animals (LD_{50} or LC_{50}, respectively).

4 48- or 96-hour LC_{50} for bluegills or rainbow trout, unless otherwise specified.

5 Registered as both insecticide and nematicide. Nematodes are controlled only on limited acreage and predominantly in the Southern states, but application rates when used as nematicides are 2- or 3-fold higher than when used as insecti cides.

6 Trade name; no corresponding common name exists.

7 24-hour LC_{50}

8 For killifish

9 For minnows

Table A-3. Agricultural fungicides: Transport modes and toxicities (from EPA-USDA Report No. EPA-600/2-75-026a, 1975)

Common Names of Fungicides	Predominant Transport Mode	Toxicity[2] Rat, Acute Oral LD_{50}, mg/kg	Fish[3] LC_{50}, mg/liter
Anilazine	S	2710	0.015
Benomyl	S	>9590	0.5
Captafol	S	5000	[4]0.031
Captan	S	9000	0.13
Carboxin	SW	3200	2.2
Chloranil	W	4000	5
Chloroneb	U	11000	>4200.0
Cycloheximide	W	2.5	1.3
DCNA	S	4040	
Dichlone	S	1300	0.047
Dichlozoline	U	3000	
Dinocap	S	980	[5]0.14
Dodine	W	1000	0.9
ETMT	U	2000	
Fenaminosulf	W	60	23
Ferbam	SW	>17000	[4]12.6
Folpet	S	>10000	[6]1.56
Maneb	S	6750	[7]1.0
Metiram	U	6400	>4.2
Nabam	W	395	[4]21.1
Ocycarboxin	W	2000	
Parinol	U	>5000	[8]~5.0
PCNB	S	1650	0.7
SMDC	W	820	[7]1.0
Thiram	S	375	[4]0.79
TPTH	U	108	
Zineb	S	>5200	0.5
Ziram	W	1400	[4]1.0

[1] Where movement of fungicides in runoff from treated fields occurs, S, denotes those chemicals that will most likely move primarily with the sediment, W denotes those that will most likely move primarily with the water, SW denotes those that will most likely move in appreciable proportion with both sediment and water. and U denotes those whose predominant mode of transport cannot be predicted because properties are unknown.

[2] Expressed as the lethal dose, or lethal concentration, to 50% of the test animals (LD_{50} or LC_{50}, respectively).

[3] 48- or 96-hour LC_{50} for bluegills or rainbow trout, unless otherwise specified.

[4] For catfish

[5] For harlequin fish

[6] For mullet

[7] LC_{100}

[8] For fathead minnow

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