AN ENVIRONMENTAL RISK INDEX
TO EVALUATE PESTICIDE PROGRAMS
IN CROP BUDGETS

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FOREWORD

The pesticide risk index presented in this paper was a part of the author's research for her Masters thesis, "Alternatives for Long Island Agriculture: The Economic Potential of Peaches and Table Grapes". The pesticide risk index is presented as a separate paper in hopes of generating discussion on how to evaluate and rank pesticides according to their potential environmental risk.

Naturally, any remaining errors are the sole responsibility of the author.

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ABSTRACT

Data on pesticide mobility, persistence, and toxicity are used to
develop a risk index which ranks pesticides according to their potential to
contaminate groundwater and surface water. The index is applied to pesti-
cide programs for several crops on Long Island and used to compare the
potential environmental risk of each.

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INTRODUCTION

Groundwater contamination from agricultural pesticides has become a major problem in recent years. On Long Island, New York, potato farmers have been forced to make major changes in their pesticide programs and cropping systems as a result of the development of Colorado Potato Beetle resistance to pesticides and the contamination of the Island's groundwater by those same pesticides.

Traditionally, farm management economists have not placed great attention upon pesticide programs in crop budgets. Most simply give a dollar value for chemicals or sometimes a breakdown into charges for fungicides, insecticides, and herbicides. However, given the increasing concern over groundwater contamination by pesticides, it seems that economists need to provide more detail on specific products and their loading rates in crop budgets.

With the increasing costs of pesticides and the frequent changes in pesticide programs, it is important that budgets specify pesticide products in detail so that subsequent changes in pesticide programs can be analyzed for their economic impact on crop production. This information also would be useful for researchers in the field of water resources as a source of "typical" crop spray programs to test in their computer simulations of pesticide leaching.

Not only is more detailed exposition of the assumptions behind chemical costs needed in crop budgets, but some assessment of the potential environmental hazard of these pesticides should be made as well. This would enable farmers and extension agents to make better informed choices of pesticide programs given the tradeoff between cost, effectiveness, and potential environmental hazard. In addition, in areas like Long Island, this kind of information is needed in order for farmers to make decisions on the ecological and economic viability of changes in crop mix.

A risk index of the potential environmental hazard from pesticides was developed for use in comparisons of pesticide programs used in crop budgets. The index was used to test the potential environmental benefits of diversification into fruits and vegetables on Long Island potato farms. This analysis was included as part of a larger study of the economic feasibility of diversification into fruit on Long Island potato farms.[1]

In this paper, a brief description of the severity of the groundwater contamination problem on Long Island is followed by a detailed description of the factors affecting pesticide behavior in soil. Past pesticide risk

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[1] For more information, the reader should refer to "Alternatives for Long Island Agriculture: The Economic Potential of Peaches and Table Grapes", M.S. Thesis, Cornell University, 1985; or A.E. Research 85-13, by M.E. Warner.

indices are critiqued, and the one developed for use in this study is presented. This index is then used to evaluate the potential environmental hazard of the spray programs for each crop considered in this study.

PESTICIDE AND NITRATE CONTAMINATION ON LONG ISLAND

Long Island has traditionally been known as a potato producing area. Although potatoes are still the largest agricultural land user on the Island, potato acreage has dropped by 50 percent in the last 13 years. The carbamate insecticide, aldicarb, was first introduced for use on Long Island in 1975. Record yields were reported in 1976 and 1977, but fell in later years as the beetle developed resistance and new, less effective chemical pesticides were substituted. In this same period (1976-1982), cost of production estimates for Long Island potato farms showed an increase of over 230 percent in the average nominal cost of pesticide use (Snyder, June 1977 and July 1982).

Even though aldicarb is highly water soluble and highly toxic, it was assumed that its toxic residues would be entirely degraded before they could reach the groundwater. However, Long Island's sandy soils allowed the insecticide to percolate quickly to the underground aquifer and contaminate the local drinking water supply.

After a preliminary survey in 1979 showed aldicarb contamination, Union Carbide asked the U.S. Environmental Protection Agency (EPA) to revoke its approval of aldicarb use on Long Island in February 1980. A full survey of 7,809 wells was carried out in 1980 and 13.1 percent were found to have aldicarb concentrations over the recommended standard of 7 ppb (Table 1). Union Carbide then supplied carbon filters to homeowners and water suppliers whose well water exceeded the standard (Baier and Moran, 1981; Baier and Robbins, 1982).

Table 1
Aldicarb Concentrations, Suffolk County, New York, 1981

Townships[a]	Number of Wells Sampled	None Detected	Less Than 7 ppb	More Than 7 ppb[b]
		- - - - - number of wells - - - - -		
Riverhead	2,161	1,465	345	351
Southold	3,160	2,427	374	359
Brookhaven	222	202	18	2
Southampton	1,832	1,306	256	270
East Hampton	434	345	46	43
Total Wells	7,809	5,745	1,039	1,025
Percent		73.6%	13.3%	13.1%

[a] See Figure 1 for location of townships in Suffolk County.
[b] New York State Department of Health drinking water standard is 7 ppb.

SOURCE: Baier, Joseph and Dennis Moran, "Status Report on Aldicarb Contamination of Groundwater as of September 1981", Suffolk County Department of Health Services.
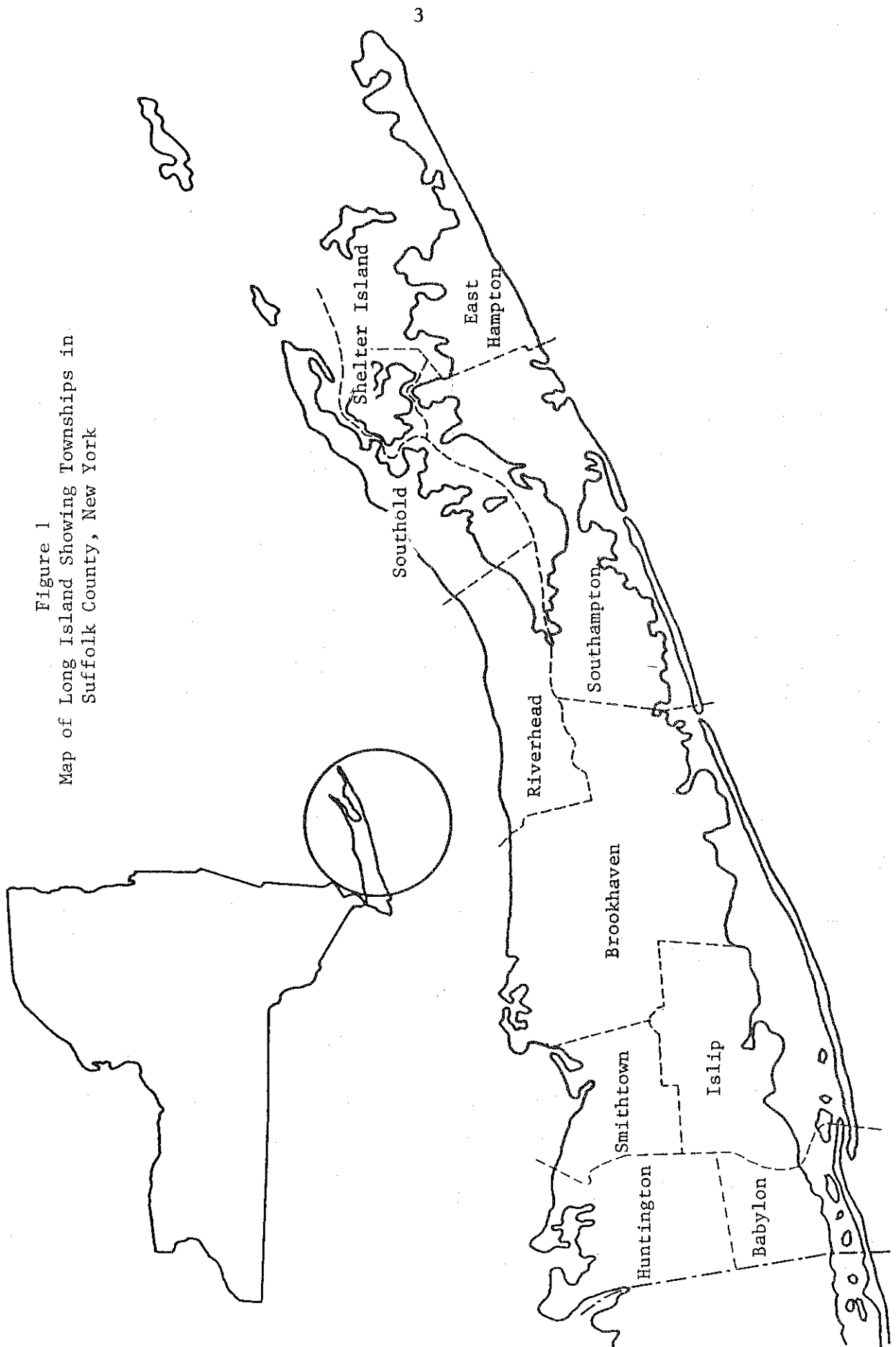
Figure 1

Map of Long Island Showing Townships in Suffolk County, New York

In 1981, the Suffolk County Health Department began its own testing program. To date they have tested for 57 materials and found 11. Of these, five have been confirmed (found more than once): aldicarb (Temik), carbofuran (Furadan), dacthal, dinoseb, and oxamyl (Vydate) (Moran, 1984).

The recent discovery of oxamyl in home wells was especially disturbing to farmers who began using it after aldicarb was banned in 1980. In January 1984, Dupont announced that it would voluntarily withdraw oxamyl from all uses on Long Island and buy back all existing stocks from suppliers on the Island. Cornell scientists and the Department of Environmental Conservation recommended use of Kryocide for the 1984 season and applied for a special permit from the EPA to use it on Long Island (Harmon, 1984). It is expected that, in the future, effective chemical controls for the Colorado Potato Beetle will become increasingly difficult to find.

Pesticides are not the only source of groundwater contamination. Nitrate concentrations above the maximum standard of 10ppm have also been found in wells on Long Island's North Fork (Table 2). The primary source of these nitrates is nitrogen fertilizer. It is estimated that applications of 200 pounds of nitrogen per acre per year in Long Island potato fields would produce an average nitrate concentration in recharged water of 16ppm. Thus, as more farmers diversify their crop production, care must be taken to limit the amount of chemical fertilizers applied as well (Baier and Robbins, 1982).

Table 2
Nitrate Concentration, Suffolk County, New York, 1981

Townships	Total	0-5ppm	5-10ppm	More than 10ppm*
		- - - - - number of wells - - - - - -		
Riverhead	639	372	167	100
Southold	1,121	575	354	192
Southampton	13,277	1,080	187	60
East Hampton	368	334	19	15
Total Wells	15,405	2,361	727	367
Percent		15.3%	4.7%	2.4%

*U.S. Public Health Service drinking water standard is 10ppm.

SOURCE: Baier, Joseph and Sy Robbins, "Report on the Occurrence and Movement of Agricultural Chemicals in Groundwater: South Fork of Suffolk County", Suffolk County Department of Health Services, August 1982.

_____, "Report on the Occurrence and Movement of Agricultural Chemicals in Groundwater: North Fork of Suffolk County", Suffolk County Department of Health Services, September 1982.

Nitrogen application rates (pounds actual) for the alternative crops considered in this study were estimated to be 175 pounds per acre for potatoes, 160 pounds per acre for cauliflower, 108 pounds per acre for peaches, 60 pounds per acre for wheat, and 50 pounds per acre for table grapes. Thus, it can be expected that diversification into fruit and rotation of potatoes with wheat will reduce the risk of nitrate contamination of groundwater.

Past studies of Long Island cropping alternatives have focused on pesticide use (Lazarus and White, 1983; Fohner, 1983). The loading rates (pounds active ingredient) of each herbicide, fungicide, and insecticide were aggregated over the farm for each crop combination. This enabled Lazarus to conclude that the substitution of vegetable and field crops for potatoes would decrease the pesticide loading rate on the farm. Fohner refined this to show that diversification would reduce use of all pesticides except organic phosphate insecticides. However, a simple accounting of loading rates falls short of answering the serious question of the impact of these pesticides on Long Island's groundwater.

Pesticide programs for all crops (potatoes, wheat, cauliflower, peaches, and table grapes) were taken from technical recommendations and adapted to reflect farmer practice. However, farmer practice varies considerably from farm to farm so the spray programs developed for these crop budgets only approximate actual pesticide use on Long Island.[2]

Pesticide contamination of groundwater on Long Island, as one of the primary motivations for this study of alternative crops for Long Island potato farmers, requires that some assessment of the environmental hazard of the pesticides be made.

PATHWAYS OF PESTICIDE LOSS

The major concern over pesticides on Long Island is the danger of groundwater contamination. Properly applied pesticides may reach surface and underground water through runoff and leaching. In order to determine the relative danger to groundwater of any pesticide, it is important to understand what can happen to a pesticide once it is applied in a field.

> Once applied to cropland, a number of things may happen to a pesticide. It may be taken up by plants or ingested by animals, insects, worms, or microorganisms in the soil. It may move downward in the soil and either adhere to soil particles or dissolve. The pesticide may vaporize and enter the atmosphere or break down via microbial and chemical pathways into other less toxic compounds. Pesticides may be leached out of the root zone by rain or irrigation water, or wash off the surface of land. The fate of a pesticide applied to soil depends largely on two of its properties: persistence and solubility (Rao, et al., 1983).

Pesticide Persistence

All pesticides applied in fields are eventually reduced from complex molecular structures to simple molecules of carbon dioxide, nitrogen, sulfur, chlorine, and various metallic elements. The time required for this "degradation" to occur varies according to the pesticide's molecular structure and the chemical, physical, and microbial characteristics of the foliage, soil, and atmosphere (Smiley, 1984). Soil microorganisms can completely break down many pesticides if they remain in the root zone (where microbe populations are highest) long enough.

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[2] Detail on products and loading rates (pounds active ingredient applied) for each crop's pesticide program are presented in Tables 8 through 14 of this report. For more detail on these pesticide programs, see A.E. Research 85-12.

Typically, degradation time is measured in half-life ($t_{1/2}$), or
the time it takes for one half the original amount of the pesticide in the
soil to be deactivated. Because half-lives of the same pesticide vary
greatly under different conditions, it is difficult to gather consistent
data on half-lives for even the most common pesticides. Mean half-life may
differ by as much as a factor of 10 between laboratory and field tests.
For this reason, some studies attempt to group pesticides in general
categories: nonpersistent, moderately persistent, and persistent (Rao, et
al., 1982; Rao, et al., 1983).

Through a review of the existing literature on the pesticides used in
these Long Island crop budgets, an attempt was made to classify each one
into a particular persistence class (Table 3).[3] The persistence
classes were: nonpersistent (half-life less than 30 days), moderately
persistent (half-life greater than 30 days but less than 100), and
persistent (half-life greater than 100 days). Where great divergence in
half-life figures occurred, pesticides were classed in accordance with
previous studies or advice from pesticide specialists at Cornell.

Table 3
Pesticide Persistence in Soils

Nonpersistent $t_{1/2}$<30 days*		Moderately Persistent 30 days <$t_{1/2}$ <100 days	
Pesticide	Mean Half-Life (days)	Pesticide	Mean Half-Life (days)
Fungicides		Fungicides	
Sulfur	Elemental	Ferbam	28 – 56
Captan	3 – 70	Maneb	28 – 56
		Mancozeb	28 – 56
		Vinclozolin	21 – 70 (like iprodione)
Insecticides		Insecticides	
Bacillus thuringiensis	0 (microbial)	Carbaryl	12 – 309
Methyl parathion	4 – 15	Diazinon	32 – 48
Parathion, ethyl	18 – 35	Piperonyl butoxide	short
Rotenone	short (botanical)	Phosmet	short
Herbicides		Herbicides	
2, 4-D	5 – 16	Dinoseb	14 – 28
		Linuron	75 – 230
		Glyphosate	38 – 903
		Metolachlor	30 – 50
		Simazine	64 – 75
		Terbacil	50 – 175

[3] In the following tables, pesticides are referred to by their common
names. See Appendix Table 1 for a list of all pesticides by common name,
product name, and chemical classification.

Table 3 continued

Pesticide	Persistent $t_{1/2} >$ 100 days	Mean Half-Life (days)
Fungicides		
Benomyl		217 - 364
Pentachloronitrobenzene		217 - 364
Metalaxyl		147 - 210
Triadamefon		217 - 364
Insecticides		
Endosulfan		>100
Herbicides		
Diuron		328
Oryzalin		240 (like trifluralin)
Paraquat		487 - 4,747
Trifluralin		28 - 544

*$t_{1/2}$ is the mean half-life in days of a pesticide

SOURCES:

Herbicide Handbook, Pub. Weed Science Society of America, Champaign, IL, 1983.

New York State College of Agriculture and Life Sciences, Pesticide Compendia, Cooperative Extension Chemicals Pesticides Program, Cornell Univ., Ithaca, NY, February 1984.

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Smiley, Richard, "Efficient Use of Fungicides on Turfgrasses", Plant Pathology Ext. Rpt. 84-1, Dept. of Plant Pathology, Cornell Univ., Ithaca, NY, January 1984.

Smith, W., Personal communication, Cooperative Extension Chemical Pesticides Program, Cornell Univ., Ithaca, NY, 1984.

Wagenet, J., Personal communication, Dept. of Agronomy, Cornell Univ., Ithaca, NY, November 1984.

Pesticide Mobility

In estimating a pesticide's mobility in soil, one of the most important properties is its solubility in water. When a pesticide enters the soil, it will interact with soil organic matter through a process called adsorption. If water enters the soil through rainfall or irrigation, the adsorbed pesticide molecules may detach from the soil particles through a process of desorption.

The adsorption partition coefficient (PC) is a useful index for measuring pesticide adsorption on soils. This index is defined as "the ratio of pesticide concentration in the adsorbed state (bound to soil particles and organic matter), and the solution-phase (dissolved in soil water)". Thus, pesticides with smaller PC values will have greater concentrations of pesticide in solution and a higher probability of leaching than those pesticides with large PC values (Rao, et al., 1983).

Partition coefficients are influenced by the solubility and melting point of pesticides. Information on partition coefficients is more readily available for herbicides and insecticides than for fungicides. Fungicides have not been subject to such studies because generally they have lower toxicity levels and it is difficult to determine how much of the product actually reaches the soil from foliar applications.[4] Since so little is known of the leaching potential of fungicides, the fungicides in this study were not rated by their adsorption characteristics.

The adsorption partition coefficient varies greatly from soil to soil. It can be normalized by soil organic carbon content according to the following equation:

$$K_{oc} = \frac{(PC)\ (100)}{\%OC}$$

Where: K_{oc} is the adsorption partition coefficient normalized by soil organic carbon.

PC is the adsorption partition coefficient.

%OC is the percentage organic carbon (weight basis) of the soil.

The variability in K_{oc} values is thereby reduced from the variability found in PC values and may be attributed to differences in the "efficiency" of the soil organic carbon among soils. Thus, K_{oc} may be a universal adsorption partition coefficient for each pesticide (Rao and Davidson, 1980, pp. 35-40).

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[4] Even the studies of herbicides and insecticides have been based on soil applications of granular or emulsifiable concentrate formulations, not foliar applications. Thus, the applicability of present data on persistence and mobility to foliar applications of herbicides and insecticides is unclear.

K_{oc} values do not exist for all pesticides. A parameter often used to estimate K_{oc} values for pesticides unstudied in soil is the octanol-water partition coefficient, K_{ow}, which can be determined in laboratory tests (Appendix Table 2). Regression analysis has shown it to have a log-linear relationship with K_{oc} for different pesticide groups. Thus, for those pesticides for which there was no reported K_{oc} value, a value was calculated using the appropriate regression equation (Rao and Davidson, 1980, pp. 40-44) (Table 4).

Several caveats must be mentioned. The coefficient of variation for many K_{oc} values is quite high (Appendix Table 3). For some pesticides, no data was available and they were classified according to field experience or K_{oc} values for pesticides with a similar chemical structure. Thus, actual mobility on Long Island soils may be quite different from what the mobility data presented here suggest.

There is a wide range in values for partition coefficients of the various pesticides. Since the adsorptive relationship between soil and pesticides is related to the clay and organic matter in the soil, soils with low clay and organic matter content will exert less retention upon pesticides. Long Island's sandy soils, with low organic matter and clay contents, constitute essentially noninteractive media in which many pesticides easily leach (Wagenet, 1984). For the purpose of this study those pesticides with PC's less than 1,000 were classified as very likely to leach.

Table 4
Partition Coefficients (K_{oc}) for Insecticides and Herbicides[a]

Immobile $K_{oc} > 1,000$		Mobile $K_{oc} < 1,000$	
Insecticides	K_{oc}	Insecticides	K_{oc}
Bacillus thurigiensis	(microbial)	carbaryl	229
Endosulfan	>1,000	diazinon	850[b]
Parathion, ethyl	10,650		
Parathion, methyl	5,101.5		
Piperonyl butoxide	(syngerist)		
Phosmet	(like parathion)		
Rotenone	(botanical)		
Herbicides		Herbicides	
Glyphosate	----	Dinoseb	151[c]
		Diuron	382.6
		Linuron	862.8
		Oryzalin	(like trifluralin)
		Simazine	138.4
		Terbacil	41.2
		Trifluralin	789[c]
		2, 4-D	19.6
		Metolachlor	----
		Paraquat	0.62[d]

[a] Partition coefficients were not available for all products. Those without numbers were classified according to the partition coefficient of their closest chemical neighbor (oryzalin and phosmet). Microbial, botanical, and synergist "pesticides" were classed as immobile because their rapid biological breakdown and low toxicity make them less dangerous to the groundwater.

Soil organic carbon partition coefficients were not available for all products. The K_{oc} was derived from the K_{ow} using the following regression equations:

[b] $\log K_{oc} = 1.029 \log K_{ow} - 0.18$ (Karickhoff and Brown in Rao and Davidson, 1980).

[c] $\log K_{oc} = 0.94 \log K_{ow} + 0.02$ (Ibid.).

[d] $\log K_{oc} = 1.0 \log K_{ow} - 0.21$ (Ibid.)

Table 4 continued

SOURCES:

Herbicide Handbook, Pub. Weed Science Society of America, Champaign, IL,
1983.

Rao, P.S.C. and J.M. Davidson, "Estimation of Pesticide Retention and
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Serv. Soil Science Fact Sheet, SL 40, Gainesville, FL, September 1983.

United States Environmental Protection Agency, Metolachlor Pesticide
Registration Standard, Office of Pesticides and Toxic Substances,
Washington, D.C., September 1980.

Wagenet, J., Personal communication, Dept. of Agronomy, Cornell Univ.,
Ithaca, NY, November 1984.

PESTICIDE TOXICITY

The concern over pesticide contamination of groundwater and surface
water stems from the concern over the toxicity of pesticides to humans,
other animals, and fish. In groundwater, the measure that is needed is the
chronic toxicity of a pesticide, that is, the cumulative effects of low
levels of exposure on human health. Unfortunately, data on chronic toxici-
ty is limited or nonexistent for many pesticides (Smith, W., 1984).[5]
Thus, often acute toxicity is used to estimate the public health risks from
contamination of groundwater sources of public drinking water.

For surface water runoff, the major concern on Long Island would be
the possible deleterious effects on the clam industry (Porter, 1984).
Changes in the salinity (due to introduction of new chemicals in the water)
and contamination (from sewage and industry) have caused many clam beds to
be closed in recent years. However, surface water runoff on Long Island is
considered to be so slight that risk to surface water from agricultural
chemicals is deemed to be practically nil.

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[5] Under the new E.P.A. registration and reregistration standards, data on
chronic toxicity and environmental fate (persistence and mobility) are
being required. Unfortunately, some of this information remains
confidential (Smith, W., 1984).

The acute toxicity, LD_{50}[6], for rats was used to classify pesticides into one of four categories:

1) relatively nontoxic: $LD_{50} > 5,000$ mg/kg;

2) slightly toxic: 500 mg/kg $\leq LD_{50} \leq 5,000$ mg/kg;

3) moderately toxic: 50 mg/kg $< LD_{50} < 500$ mg/kg;

4) highly toxic: $LD_{50} \leq 50$ mg/kg;

(New York State Pesticide Recommendations, 1983). See Table 5 for toxicity classifications of the pesticides used in this study.

ESTIMATING ENVIRONMENTAL RISK

Indices Used in Past Studies

Some studies of field crops have attempted to measure the environmental impact of pesticides by evaluating the tradeoff between toxicity and persistence in the environment (Gould, 1983; Alt, 1976). According to Alt (p.38) an "ideal insecticide would persist only until all target organisms are controlled and would be nontoxic to all other organisms." He described three possible indices which provide a common denominator to measure this tradeoff between toxicity and persistence. First was the Potential Environmental Hazard (PEH) index (Weber, in Alt, 1976) which was based on four factors: 1) mobility (M), 2) longevity (L), 3) toxicity (T), and 4) biomagnification (B). Each pesticide was assigned a rating and the ratings were combined by multiplication to give the pesticide an index value (PEH = M • L • T • B).

The second index was the Environmental Harm Coefficient (EHC) (Dixon, Dixon, and Miranowski, in Alt, 1976) which was based upon insecticide toxicity and rate of decomposition. The value of the index was determined by dividing the pounds active ingredient (AI) of a pesticide by the product of the decomposition rate (d) and the toxicity (LD_{50}) and summing over

all pesticides. $$(EHC = \sum_{j=i}^{n} \frac{AI_j}{d_j \cdot LD_j})$$

These indices only considered insecticides and failed to differentiate insecticides of the same chemical group since persistence values were often assumed to be the same for all insecticides of a particular group (e.g., organophosphates). Alt attempted to improve on these previous indices by using a unique measure of persistence for each insecticide and a first-order kinetic degradation function to derive an estimate of environmental exposure. The value of his Environmental Exposure Index (EEI) was determined by dividing the product of the half-life (h) and application

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[6] The LD_{50} is the lethal dose in miligrams per kilogram of bodyweight at which half the population (usually of rats) died.

Table 5
Pesticide Toxicity

Relatively Nontoxic		Slightly Toxic		Moderately Toxic		Highly Toxic	
LD_{50} >5000 mg/kg		500 mg/kg $\leq LD_{50} \leq$ 5000mg/kg		50 mg/kg < LD_{50} <500mg/kg		0 mg/kg < LD_{50} $\leq$ 50mg/kg	
Product	LD_{50}	Product	LD_{50}	Product	LD_{50}	Product	LD_{50}
Fungicides							
Benomyl	>10,000	Ferbam	>4000	Triadamefon	363-568		
Captan	9000	Metalaxyl	669				
Mancozeb	>8000						
Maneb	6750						
Pentachloro-nitrobenzene	>12,000						
Vinclozolin	10,000						
Sulfur	nontoxic						
Insecticides							
Bacillus thuringiensis	harmless	Carbaryl	500-850	Diazinon	300-400	Ethyl parathion	3.6-13
Piperonyl butoxide	>7500			Endosulfan	70-110	Methyl parathion	14-24
				Phosmet	113-350		
				Rotenone	132-1500		
Herbicides							
Oryzalin	>10,000	Diuron	3400	2, 4-D	375-800	Dinoseb	40-58
Simazine	>5000	Glyphosate	4300-5600	Paraquat	150-157		
Terbacil	>5000	Linuron	1500-4000				
Trifluralin	>10,000	Metolachlor	2780				

SOURCES:
New York State College of Agriculture and Life Sciences, Pesticide Compendia, Cooperative Extension Chemicals
Pesticides Program, Cornell Univ., Ithaca, NY, February 1984.
University of Maryland, "Pesticide Profiles; Part One: Insecticides and Miticides", Cooperative Extension
Service Bul. 267, College Park, MD, 1979.
Worthing, Charles R., The Pesticide Manual, British Crop Protection Council, Lavenham, Suffolk, England, 1983.

rate (A) of an insecticide by the product of the toxicity (LD_{50}) and

the natural logarithm of 2. $$\left(EEI = \sum_{j=1}^{n} \frac{h_j \cdot A_j}{\ln 2 \cdot LD_j} \right)$$

Although some studies have assumed that degradation follows a first order kinetic function, the exact function describing chemical degradation is not known. These indices, by suggesting a functional form for pesticide environmental fate when none is known, give a false sense of accuracy. They also fail to differentiate pesticides by their potential to contaminate groundwater or surface water. Most studies of environmental hazard now take the form of computer simulations which include more of the variables which are known to affect the chemical in the environment.

While computer simulations give a more accurate estimate of groundwater contamination, a simpler method of evaluating the environmental fate of pesticides is needed. Pesticide contamination is of interest in many studies where computer simulations would not be justified. A simple method of ranking pesticides according to their environmental hazard would enable more researchers to include pesticide contamination evaluations in their studies of crop production.

Ranking Pesticides by Environmental Risk

A simple ranking system was developed to evaluate the potential environmental hazard of the pesticide programs used in the crops included in this study. The pesticide persistence, mobility, and toxicity classifications (Tables 3, 4, and 5), were used to develop the ranking system. Each pesticide was assigned a risk index value reflecting the tradeoff between its likelihood to contaminate and its toxicity.

Likelihood was defined as the mobility (partition coefficient) and persistence (half-life) of a pesticide which characterize its ability to contaminate surface or groundwater. Those pesticides with high partition coefficients (greater than 1,000) were assumed to be relatively immobile and more likely to contaminate surface water. Those pesticides with partition coefficients under 1,000 were assumed to be relatively more mobile and more of a threat to groundwater.

In the case of surface water, persistence was used to determine the degree of likelihood, with more persistent pesticides having a higher likelihood rating. Since soil persistence gives little indication of persistence in a nonbiologically active environment (such as groundwater), all those pesticides which were classified as mobile were assumed to have a high likelihood to contaminate groundwater (Lemley, 1984). Table 6 illustrates the likelihood ratings.

Table 6
Pesticide Likelihood to Contaminate Surface or Groundwater

	Mobility	
Persistence	Immobile	Mobile
Nonpersistent	Low	High
Moderately persistent	Moderate	High
Persistent	High	High
Destination	Surface Water	Groundwater

These likelihood ratings were then combined with the toxicity ratings
to build a matrix of risk index values. The standard EPA definition of
risk, "the probability that a substance will produce harm under specified
conditions", was interpreted as a function of likelihood to contaminate
surface or groundwater and acute toxicity. Some would argue that this
function could be expressed as a multiplicative (toxicity rating x likeli-
hood rating) or additive (toxicity rating + likelihood rating) relationship
(Porter, 1984). Although the neatness of such a "mathematical" approach is
attractive, it does not do as good a job of ranking the various toxicity
combinations against each other as the matrix presented in Table 7.

Table 7
Risk Index Values For Pesticides

	Toxicity Rating			
Likelihood Rating	Nontoxic	Slightly Toxic	Moderately Toxic	Highly Toxic
Surface Water				
Low	1	2	5	6
Moderate	1	3	7	8
High	1	4	9	10
Groundwater				
High	1	4	9	10

The risk index matrix was based on assumptions from experts of the
relative ranking of various likelihood/toxicity combinations (Lemley, 1984;
Smith, 1984; and Wagenet, 1984). The matrix is divided into three regions
of increasing risk. The least risky is the region of the two lowest
toxicity ratings. At the lowest toxicity rating, it is assumed that
likelihood to contaminate does not matter. The products are relatively
nontoxic, so they pose the lowest potential risk to the environment. For
those products in the slightly toxic category, likelihood to contaminate
does matter and the risk rankings given these products rise with the
increase in likelihood to contaminate (2,3,4).

The right half of the matrix, composed of the two highest toxicity
ratings, represents the area of greatest risk. Here, likelihood becomes

even more important than toxicity (contrary to the case in column two). It is felt that, at these higher toxicity ratings, the danger of a higher likelihood to contaminate is more important than the danger of higher toxicity.

This half of the matrix can be divided into two regions of increasing risk. The region of low likelihood to contaminate (ratings 5 and 6) is less risky than the region of moderate and high likelihood. Any pesticide given a ranking of 7 to 10 should be viewed with extreme caution since it poses the greatest potential environmental risk.

The numerical rankings express only <u>qualitative</u> differences in the risk of particular pesticides. The index numbers, in and of themselves, are meaningless; i.e., a risk index of 4 does not mean that the product is four times more risky than a pesticide with an index of 1. No attempt is made to suggest the exact functional relationship of relative pesticide risk. The index simply attempts to rank the potential risk of one pesticide vis-a-vis another.

RELATIVE RISK RANKINGS OF CROP PESTICIDE PROGRAMS

Deriving Pesticide Risk Rankings

Risk index values were determined for each pesticide. Fungicide, herbicide, and insecticide spray programs for each crop were considered separately. Each product in a crop's spray program was used to form a weighted average risk index for that spray program according to the following equation:

$$\overline{RI} = \sum_{j=1}^{n} \frac{RI_j \ AI_j}{AI_j}$$

where: $\overline{RI}$ = the weighted average risk index for the fungicide, herbicide, or insecticide program for a particular crop.

RI_j = the risk index for a particular pesticide.

AI_j = pounds active ingredient of a particular pesticide.

j = each pesticide.

The loading rate [sum of pounds active ingredient ($\sum_{j=1}^{n} AI_j$)] and the weighted average risk index ($\overline{RI}$) were determined separately for potential groundwater and surface water contaminators for each crop spray program. A distinction was not made between groundwater and surface water contamination from fungicides since too little is known about the partition coefficients of fungicides to predict their mobility.

Tables 8 through 14 show the derivations of the risk index rankings for the fungicide, insecticide, and herbicide programs for potatoes as well as the alternative crops of wheat, cauliflower, peaches, and table grapes.

Table 8
Potato Pesticide Risk Rankings

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
FUNGICIDES					
Maneb	1.68	NA	moderate	non	1
Mancozeb	16.15	NA	moderate	non	1
Metalaxyl	1.74	NA	high	slight	4
	$\Sigma AI_j = 19.57$			$\Sigma RI_j \ AI_j = 24.79$	
				$\overline{RI} = 1.27$	
INSECTICIDES (continuous potatoes)					
Endosulfan	5.00	surface	high	moderate	9
Parathion	5.00	surface	low	high	6
Rotenone	1.30	surface	low	moderate	5
Piperonyl butoxide	2.50	surface	moderate	non	1
Surface	$\Sigma AI_j = 13.80$			Surface $\Sigma RI_j \ AI_j = 84.0$	
				Surface $\overline{RI} = 6.09$	
INSECTICIDES (rotated potatoes)					
Endosulfan	4.00	surface	high	moderate	9
Parathion	4.00	surface	low	high	6
Rotenone	1.04	surface	low	moderate	5
Piperonyl butoxide	2.00	surface	moderate	non	1
Surface	$\Sigma AI_j = 11.04$			Surface $\Sigma RI_j \ AI_j = 67.2$	
				Surface $\overline{RI} = 6.09$	
HERBICIDES					
Metolachlor	1.51	ground	high	slight	4
Linuron	1.00	ground	high	slight	4
Dinoseb	3.00	ground	high	high	10
Ground	$\Sigma AI_j = 5.51$			Ground $\Sigma RI_j \ AI_j = 40.04$	
				Ground $\overline{RI} = 7.27$	

Table 9
Cauliflower Pesticide Risk Rankings

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
FUNGICIDES					
Captan	0.35	NA	low	non	1
Pentachloro- nitrobenzene	0.52	NA	high	non	1
Maneb	4.95	NA	moderate	non	1
$\Sigma AI_j =$ 5.82				$\Sigma RI_j\ AI_j =$ 5.82	
				$\overline{RI}$ = 1.0	
INSECTICIDES					
Diazinon	1.18	ground	high	moderate	9
Endosulfan	3.00	surface	high	moderate	9
Bacillus thuringiensis	1.50	surface	low	non	1
Ground $\Sigma AI_j =$ 1.18				Ground $\Sigma RI_j\ AI_j =$ 10.62	
				$\overline{RI}$ = 9.0	
Surface $\Sigma AI_j =$ 4.50				Surface $\Sigma RI_j\ AI_j =$ 28.5	
				$\overline{RI}$ = 6.33	
HERBICIDES					
Trifluralin	1.03	ground	high	non	1
Ground $\Sigma AI_j =$ 1.03				Ground $\Sigma RI_j\ AI_j =$ 1.03	
				Ground $\overline{RI}$ = 1.0	

Table 10
Wheat Pesticide Risk Rankings

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
HERBICIDES					
2, 4-D	0.50	ground	high	moderate	9
Ground $\Sigma AI_j =$ 0.50			Ground	$\Sigma RI_j \, AI_j =$	4.5
				$\overline{RI} =$	9.0

Table 11
Table Grape Pesticide Risk Rankings
(Years One and Two)

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
FUNGICIDES					
Mancozeb	7.60	NA	moderate	non	1
Sulfur	7.60	NA	low	non	1
Triadamefon	0.12	NA	high	moderate	9
$\Sigma AI_j =$ 15.32				$\Sigma RI_j \, AI_j =$	16.28
				$\overline{RI} =$	1.06
INSECTICIDES					
Carbaryl	2.00	ground	high	slight	4
Ground $\Sigma AI_j =$ 2.00			Ground	$\Sigma RI_j \, AI_j =$	8.0
				$\overline{RI} =$	4.0
HERBICIDES					
Oryzalin	0.75	ground	high	non	1
Paraquat	0.08	ground	high	moderate	9
Ground $\Sigma AI_j =$ 0.83			Ground	$\Sigma RI_j \, AI_j =$	1.47
				$\overline{RI} =$	1.77

Table 12
Table Grape Pesticide Risk Rankings
(Mature Vineyard)

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
FUNGICIDES					
Ferbam	3.42	NA	moderate	slight	3
Captan	3.00	NA	low	non	1
Sulfur	11.40	NA	low	non	1
Mancozeb	11.20	NA	moderate	non	1
Maneb	5.60	NA	moderate	non	1
Triadamefon	0.25	NA	high	moderate	9
Benomyl	1.00	NA	high	non	1
Vinclozolin	1.50	NA	moderate	non	1

$$\Sigma AI_j = 37.37 \qquad \Sigma RI_j\, AI_j = 46.21$$
$$\overline{RI} = 1.24$$

INSECTICIDES					
Carbaryl	8.00	ground	high	slight	4

$$Ground \quad \Sigma AI_j = 8.00 \qquad Ground \quad \Sigma RI_j\, AI_j = 32.00$$
$$\overline{RI} = 4.0$$

HERBICIDES					
Glyphosate	0.25	surface	moderate	slight	3
Oryzalin	0.75	ground	high	non	1
Simazine	0.48	ground	high	non	1

$$Ground \quad \Sigma AI_j = 1.23 \qquad Ground \quad \Sigma RI_j\, AI_j = 1.23$$
$$\overline{RI} = 1.0$$
$$Surface \quad \Sigma AI_j = 0.25 \qquad Surface \quad \Sigma RI_j\, AI_j = 0.75$$
$$\overline{RI} = 3.0$$

Table 13
Peach Pesticide Risk Rankings
(Years One and Two)

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
FUNGICIDES (Years 1 & 2)					
Ferbam	1.14	NA	moderate	slight	3
Captan	3.00	NA	low	non	1
Sulfur	6.00	NA	low	non	1
$\Sigma AI_j =$ 10.14				$\Sigma RI_j \, AI_j =$ 12.42	
				$\overline{RI} =$ 1.22	
INSECTICIDES (Year 1)					
Parathion, methyl	0.45	surface	low	high	6
Surface $\Sigma AI_j =$ 0.45				Surface $\Sigma RI_j \, AI_j =$ 2.7	
				$\overline{RI} =$ 6.0	
INSECTICIDES (Year 2)					
Parathion, methyl	0.60	surface	low	high	6
Surface $\Sigma AI_j =$ 0.60				Surface $\Sigma RI_j \, AI_j =$ 3.6	
				$\overline{RI} =$ 6.0	
HERBICIDES (Year 1)					
Oryzalin	0.75	ground	high	non	1
Dinoseb	1.00	ground	high	high	10
Ground $\Sigma AI_j =$ 1.75				Ground $\Sigma RI_j \, AI_j =$ 10.75	
				Ground $\overline{RI} =$ 6.14	
HERBICIDES (Year 2)					
Simazine	0.33	ground	high	non	1
Oryzalin	0.75	ground	high	non	1
Paraquat	0.15	ground	high	moderate	9
Ground $\Sigma AI_j =$ 1.23				Ground $\Sigma RI_j \, AI_j =$ 2.43	
				$\overline{RI} =$ 1.98	

Table 14
Peach Pesticide Risk Rankings
(Mature Orchard)

Product	Active Ingredient Lbs. (AI)	Water Contamination	Likelihood	Toxicity	Risk Index (RI)
FUNGICIDES					
Ferbam	1.44	NA	moderate	slight	3
Benomyl	1.30	NA	high	non	1
Captan	12.50	NA	low	non	1
Sulfur	27.56	NA	low	non	1
	$\Sigma AI_j = 42.80$			$\Sigma RI_j\ AI_j = 45.68$	
				$\overline{RI} = 1.07$	
INSECTICIDES					
Endosulfan	2.13	surface	high	moderate	9
Parathion, methyl	1.20	surface	low	high	6
Phosmet	1.25	surface	moderate	moderate	7
Carbaryl	4.00	ground	high	slight	4
Ground	$\Sigma AI_j = 4.00$			Ground $\Sigma RI_j\ AI_j = 16.0$	
				$\overline{RI} = 4.0$	
Surface	$\Sigma AI_j = 4.58$			Surface $\Sigma RI_j\ AI_j = 35.12$	
				$\overline{RI} = 7.67$	
HERBICIDES					
Simazine	0.33	ground	high	non	1
Oryzalin	0.75	ground	high	non	1
Paraquat	0.15	ground	high	moderate	9
Ground	$\Sigma AI_j = 1.23$			Ground $\Sigma RI_j\ AI_j = 2.43$	
				$\overline{RI} = 1.98$	

Comparison of Pesticide Risk Rankings by Crop

A summary of pesticide loading rates (ΣAI_j) in each crop and of the weighted average risk index ($\overline{RI}$) showed that although the amount of fungicides applied in mature peaches and grapes was approximately twice as large as that applied in potatoes, the risk index was almost the same (Table 15). For insecticides, the danger to groundwater from peaches and grapes was actually higher than from potatoes because the mobile pesticides previously used in potatoes (aldicarb and oxamyl) have now been banned. However, the insecticide which caused peaches and grapes to show a higher risk of contaminating groundwater, carbaryl, could easily be replaced with a product such as parathion which does not pose a threat to groundwater.

Table 15
Comparison of Pesticide Programs by Loading Rate and
Risk Index, All Crops, Long Island

| | Fungicides | | Insecticides | | | | Herbicides | | | |
| | | | Ground-water | | Surface water | | Ground-water | | Surface water | |
Product	ΣAI_j	$\overline{RI}$	ΣAI_j	$\overline{RI}$	ΣAI_j	$\overline{RI}$	ΣAI_j	$\overline{RI}$	ΣAI_j	$\overline{RI}$
Potatoes										
Continuous	19.57	1.27	0.00	0.00	13.80	6.09	5.51	7.27	0.00	0.00
Rotated	19.57	1.27	0.00	0.00	11.04	6.09	5.51	7.27	0.00	0.00
Cauliflower	5.82	1.00	1.18	9.00	4.50	6.33	1.03	1.00	0.00	0.00
Wheat	0.00	0.00	0.00	0.00	0.00	0.00	0.50	9.00	0.00	0.00
Peaches										
Year 1	10.14	1.22	0.00	0.00	0.45	6.00	1.75	6.14	0.00	0.00
Year 2	10.14	1.22	0.00	0.00	0.60	6.00	1.23	1.98	0.00	0.00
Years 3+	42.80	1.07	4.00	4.00	4.58	7.67	1.23	1.98	0.00	0.00
Grapes										
Years 1 & 2	15.32	1.06	2.00	4.00	0.00	0.00	0.83	1.77	0.00	0.00
Years 3+	37.37	1.24	8.00	4.00	0.00	0.00	1.23	1.00	0.25	3.00

ΣAI_j = loading rate, total pounds active ingredient applied.

$\overline{RI}$ = weighted average risk index.

Herbicides posed the greatest threat to groundwater in potatoes where dinoseb is used as a vinekiller. In fact, dinoseb has been found in surveys of wells on Long Island. Peaches in the establishment year posed a similar threat, but in the mature orchard the threat was reduced since dinoseb was no longer used.

Although surface water contamination from insecticides in potatoes and peaches appeared high, the low danger of surface runoff on Long Island caused these risk indices to be of little concern.

Thus, it appears that a move into peaches and grapes may not pose a threat to Long Island's groundwater despite the heavy spray programs which production of these crops implies. Naturally, it is impossible to predict whether the pesticides used in peaches and grapes will actually contaminate the groundwater on Long Island. However, this analysis of the characteristics of the fruit pesticide programs has offered a clue to the potential danger of groundwater contamination. For fruit production, estimation of this danger is important because of the difficulty of changing to other crops once the orchards and vineyards have been established.

IMPACT OF CULTURAL PRACTICES ON PESTICIDE CONTAMINATION

Although chemical characteristics of pesticides can provide insight into possible groundwater problems, cultural practices may be even more important as determining factors. Danger of groundwater contamination can be reduced through buildup of soil organic matter and more frequent applications of smaller doses of pesticides, which will reduce leaching from rainfall to a minimum. Irrigating prior to and not immediately after a pesticide application will also help. However, on Long Island's porous soils, leaching will always be a problem with any pesticide with a small partition coefficient. Careful irrigation and maintenance of sod row middles in orchards and vineyards would help decrease the danger of runoff and surface water contamination. Sod ground cover would also help maintain a biologically active root zone to degrade the pesticides (Wagenet, 1984).

Farmers on Long Island must always be cautious in their pesticide use since the characteristics of Long Island's soil and underground aquifer make it an especially fragile environment. No matter which crops are grown, the danger of pesticide contamination will persist. The risk rankings presented here should be treated as no more than an educated guess of the potential hazard to Long Island's groundwater.

High inputs of pesticides have been assumed in this study to reflect complete insect and disease control. With careful focus on integrated pest management schemes, farmers could reduce pesticide use from the levels predicted here and thereby reduce the risk of groundwater contamination. Crop diversification and rotation also should help to reduce pest populations and, at the very least, spread the risk from future pesticide bans.

CONCLUSION

As farmers, extension agents, and economists seek to find ways to reduce pesticide risk to groundwater through production of new crops or use of new pesticides, they must be careful not to make the fallacious assumption that lower application rates are intrinsically better. This was the underlying assumption in the studies by Fohner and Lazarus and, although it seems logical, it fails to take into consideration the relative risk of one pesticide vis-a-vis another.

This paper has presented a risk index which provides a way to make qualitative judgements about the potential environmental risk of particular pesticides and crop spray programs. For example, even though mature peaches and table grapes used more fungicides than potatoes, the risk index showed that, in terms of potential environmental risk, they were no worse than potatoes. A simple accounting of loading rates would have reached the opposite conclusion.

Clearly this simple ranking scheme cannot accurately predict relative pesticide risk in all cases. Given the scarcity of data on field behavior of pesticides, one cannot say with certainty what the fate of pesticides applied in the field will be. However, until such time as more field data are collected and computer simulations made more readily available to practitioners, some simple ranking scheme is needed to aid decision makers in choosing alternative crops or pesticides. This is especially important in ecologically fragile areas like Long Island. The risk index presented here is a rough, first attempt at such a ranking scheme.

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Appendix Table 1
COMMON NAMES, PRODUCT NAMES, AND CHEMICAL CLASSIFICATIONS OF PESTICIDES

Common Name	Product Name	Chemical Classification
FUNGICIDES		
Benomyl	Benlate	Carbamate
Captan	Captan	Dicarboximide
Ferbam	Ferbam	Dithiocarbamate
Mancozeb	Dithane M45	Dithiocarbamate
	Manzate 200	
Maneb	Dithane M-22	Dithiocarbamate
	Manzate	
Metalaxyl	Ridomil	Acylalamine compound
Pentachloronitrobenzene	P.C.N.B.	Chlorinated Hydrocarbon
	Terraclor	
Sulfur	Sulfur	Inorganic element
Triadamefon	Bayleton	Organic
Vinclozolin	Ronilan	Organic
INSECTICIDES		
Bacillus Thuringiensis	Dipel	Microbial Insecticide
Carbaryl	Sevin	Carbamate
Diazinon	Diazinon	Organophosphate
Endosulfan	Thiodan	Chlorinated Hydrocarbon
Parathion, ethyl	Folidol, Fosferno	Organophosphate
Parathion, methyl	Folidol-M, Penncap-M	Organophosphate
Phosmet	Imidan	Organophosphate
Piperonyl butoxide	P.B.O.	Synergist
Rotenone	Rotenone	Botanical Insecticide
HERBICIDES		
2, 4-D	2, 4-D	Phenoxy Compound
Dinoseb	Premerge 3	Dinitro
Diuron	Karmex	Substituted Urea
Glyphosate	Roundup	Glycine
Linuron	Lorox	Substituted Urea
Metolachlor	Dual	Acetamide
Oryzalin	Surflan	Dinitroanaline
Paraquat	Paraquat	Bypyridylium
Simazine	Princep	Triazine
Terbacil	Sinbar	Uracil
Trifluralin	Treflan	Dinitroanaline

Appendix Table 2
OCTANOL-WATER PARTITION COEFFICIENTS FOR PESTICIDES

Pesticide	K_{ow}	Log K_{ow}
Carbaryl	6.51000E+02	2.81358E+00
Diazinon	1.05200E+03	3.02202E+00
Dinoseb	1.98000E+02	2.29667E+00
Methyl Parathion	2.07600E+03	3.31723E+00
Parathion	6.45500E+03	3.80990E+00
Diuron	6.50000E+02	2.81291E+00
Paraquat .2HCL	1.00000E+00	0.00000E+00
Simazine	8.80000E+01	1.94448E+00
Terbacil	7.80000E+01	1.89209E+00
Trifluralin	1.15000E+03	3.06070E+00
2, 4-D	4.43000E+02	2.64640E+00

Appendix Table 3
ADSORPTION PARTITION COEFFICIENTS FOR PESTICIDES

Pesticide	Number of Soils	K_{oc} (mean)	(%CV)
Diuron	84	382.6	(72.4)
Linuron	33	862.8	(72.3)
Methyl Parathion	7	5,101.5	(113.6)
Parathion	4	10,650.3	(74.6)
Simazine	147	138.4	(12.6)
Terbacil	4	41.2	(42.2)
2, 4-D	9	19.6	(72.4)

SOURCE: Rao, D.S.C. and J.M. Davidson, "Estimation of Pesticide Retention and Transformation Parameters Required in Nonpoint Source Pollution Models", in Overcash, M.R. and Davidson, J.M., eds., Environmental Impact of Nonpoint Source Pollution, Ann Arbor Science Pub., Inc., Ann Arbor, MI, 1980, pp37-39, (Table 2) and pp41-43 (Table 3).