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Agriculture, Ecosystems and Environment, 9 (1983) 359–372
Elsevier Science Publishers B.V., Amsterdam — Printed in The Netherlands

ENERGY EFFICIENCY OF FARMING SYSTEMS: ORGANIC AND CONVENTIONAL AGRICULTURE

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(Accepted 31 January 1983)

ABSTRACT

Pimentel, D., Berardi, G. and Fast, S., 1983. Energy efficiency of farming systems: organic and conventional agriculture. *Agric. Ecosystems Environ.*, 9: 359–372.

An assessment was made of the energy efficiency, yield performance, and labor requirements for the production of corn, wheat, potatoes, and apples using organic (without synthetic chemical fertilizers and pesticides) and conventional farming technologies. Organic corn and wheat production was 29–70% more energy efficient than conventional production. However, conventional potato and apple production was 7–93% more energy efficient than organic production. For all four crops, the labor input per unit of yield was higher for organic systems compared with conventional production.

INTRODUCTION

Since the mid 1930's, agricultural productivity measured in crop yield per acre has more than doubled (USDA, 1980). The United States now dominates the world's grain exports and in the fiscal year 1981, U.S. agricultural exports are projected to reach a record $45 billion. This high level of productivity has been due largely to the mobilization of energy resources in agricultural production combined with the use of high-yielding crop varieties and in part to timeliness of operations and other cultural practices (Jensen, 1978).

The large fossil energy subsidies needed to maintain the U.S. agricultural system have been the subject of much research (Pimentel, 1980). The growing interest over the magnitude of the energy inputs is shared not only by researchers but also by individual farmers who are trying to minimize energy inputs and thus production costs (Berardi, 1976; Wernick and Lockeretz, 1977), and by consumers who may ultimately pay higher food prices (Steinhart and Steinhart, 1974; Leach, 1976).

The amount of energy expended for food production, distribution, and preparation in the United States represents about 17% of total U.S. energy;

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0167-8809/83/$03.00

approximately one-third (6%) of this is used for food production (Pimentel and Pimentel, 1979). The major fossil energy inputs to the agricultural system are fuel used for machinery operations and synthetic fertilizers (Pimentel, 1980). Methods of reducing fuel use on the farm have been the subject of much investigation. There are many opportunities for reducing energy inputs in crop production (Pimentel et al., 1973; Berardi, 1978; Pimentel, 1980; Lockeretz et al., 1981). Reduction in the use of synthetic fertilizers, for example, could result in significant energy savings in agriculture. The production of fertilizers, primarily nitrogenous, requires nearly 30% of the total energy expended in U.S. crop production (Pimentel, 1980).

Given the high energy requirement, monetary cost, and environmental cost of synthetic fertilizers, as well as pesticides, research efforts are focused on the optimal use of fertilizers and pesticides and on seeking alternative technologies. Organic agriculture is frequently suggested as one alternative technology. This study is an assessment of the energy efficiency, yield performance, and labor requirements of organic agricultural technologies compared with conventional agricultural production. Comparisons are made for four crops: corn, wheat, potatoes, and apples.

GENERAL METHODS

In this study organic farming is defined as a production system that avoids or excludes the use of synthetic chemical fertilizers, pesticides, and growth regulators (USDA, 1980). The essential crop nutrients of nitrogen (N), phosphorus (P), and potassium (K), are provided by crop residues, livestock manure, legumes used as a nitrogen source (e.g., soybeans and sweet clover), and off-farm organic wastes such as sewage sludge and other soil amendments (glauconite (K), rock phosphate (P)). Both organic and conventional farmers employ a range of farming techniques, including choices in crops, types of tillage, crop diversity, and presence or absence of livestock (Oelhaf, 1978; USDA, 1980).

For the organic systems that required manure and sewage sludge in this study, it was assumed that these nutrient sources were reasonably close to the cropping area. Clearly, manure and sewage sludge with about 85% water could not be efficiently transported by tractor or truck more than 10–30 km. For this example, we assume a 3 km distance for the livestock manure source and 15 km for the sewage sludge. We also assume that the manure is transported and spread by tractor with an input of 15 000 kcal (ca. 0.5 gallon of fuel) per tonne of manure (Linton, 1968), and that the sewage sludge is transported and spread by a truck with an input of 15 000 kcal t^{-1}. This efficiency is based on the assumption that the truck can be driven on the cropland.

The cattle manure is calculated to contain 5.6 kg of N, 1.5 kg of P, and 3 kg of K per wet tonne (Pimentel et al., 1973). Concerning the availability of N from fresh manure, about half is in the form of ammonia and half as organic N. During the first growing season 80% of the ammonia is available to the crop and 40% of the organic N is mineralized and available (R.E. Muck, personal communication, 1982). If manure is applied each year, then earlier applications continue to be mineralized until an equilibrium is reached. At this time the amount of N mineralized and potentially available to the crop is about 75% of the annual application. Of course, a small percentage of commercial N is also lost and not available to the crop.

The sludge is calculated to contain 3.0 kg of N, 1.1 kg of P, and 0.8 kg of K per wet tonne (Metcalf and Eddy, 1972). The availability of N from sludge is assumed to be similar to that for livestock manure (Magdoff and Amadon, 1980).

In those cases where P is needed in addition to the organic waste material, rock phosphate is used. The effectiveness of rock phosphate depends on soil pH and the concentration of P and Ca in the soil solution (USDA, 1980). Although rock phosphate releases P more slowly than acid-treated phosphate, it was assumed that the material had been used for several years and an adequate equilibrium had been established (Lockeretz, 1980; USDA, 1980). Therefore, nutrient availability was not a problem (USDA, 1980). The calculated energy input for rock phosphate was 1300 kcal kg^{-1} (Lockeretz, 1980). Similarly, additional K was supplied by low-solubility sources of K such as glauconite (green sand) (USDA, 1980). The energy input for glauconite was 2200 kcal kg^{-1} (Berardi, 1976). In all cases, we assumed that the organic approaches of supplying nutrients were adequate for crop needs.

For pest control, non-chemical control alternatives were limited primarily to weed control for the four crops used in the analysis. The substitute technology for herbicidal weed control was additional mechanical cultivation. For insect control, the only readily available alternative non-chemical controls were crop rotations for the control of the corn rootworm complex and host plant resistance and planting time for the control of the Hessian fly of wheat (Pimentel et al., 1982). However, for most insect pests in this study, we assumed that there were no effective non-chemical controls at present. Also, for all plant pathogens, we assumed there were no non-chemical controls available for the four crops. Of the four crops selected for this analysis two crops, corn and wheat, can be produced with minimal pest losses whereas both potatoes and apples suffer severe losses from insects and plant pathogens if pesticides are not employed (Pimentel et al., 1978). Crop locations were chosen to represent typical crop producing regions.

No attempt was made in this study to substitute different types of tillage for moldboard plow or machinery for basic field operations for either organic or conventional agricultural practices. Energy and labor costs of plowing down previous crops in the organic rotation were considered part of both organic and conventional tillage operations unless the crop was grown only as a green manure for organic production. Labor requirements also remained the same for basic tillage and harvesting operations for both conventional

and organic production. Additional labor was needed to appply livestock manure or sewage sludge.* Another additional cost for organic crop production was extra labor hours for weed control. No additional labor was included for spreading rock phosphate and glauconite since this is similar to the labor required to apply chemical fertilizers on conventional farms.

ORGANIC AND CONVENTIONAL CROP PRODUCTION

Corn

For this analysis of corn produced under organic and conventional agricultural production technologies, energy, labor, and yield data were used for corn grown in Iowa (Pimentel and Burgess, 1980). Input data for conventional corn production are listed in Table I. For organic corn production the combinations examined were: livestock manure; digested sewage sludge; corn after alfalfa; corn after soybeans; and corn after a sweet clover fallow (Table I).

The yield of corn grown by conventional agricultural technology was updated to 8005 kg ha^{-1} (ca. 128 bushels per acre, the average yield in Iowa in 1981 (USDA, 1981)) with an energy value of 27.9 × 10^6 kcal and a labor input of 9.6 h (Table I). Thus, the energy production ratio, i.e., the ratio of kcal output per kcal input, was 4.47 and the labor productivity was 834 kg of corn produced per hour.

All corn grown organically was assumed to be planted after a crop other than corn so that corn rootworm was not a problem. By using crop rotation without insecticides, insect losses were estimated to increase by 1% (Pimentel et al., 1978)**. Herbicide use was assumed to be replaced by two additional cultivations, with a labor input of 1 h ha^{-1} and an energy input of 14 l of diesel fuel ha^{-1} per cultivation, or a total of 91 300 kcal (Lockeretz et al., 1976). Raising organic corn using cattle manure as the major source of nutrients is calculated to produce 7925 kg ha^{-1} with an energy value of 27.6 × 10^6 kcal and a labor input of 14.8 h (Table I). Thus, the energy production ratio is 7.34 or 64% better than conventionally grown corn. However, the labor productivity was only 534 kg h^{-1} or 35% poorer than conventional corn production (Table I).

*These additional crop labor costs might be calculated as zero since spreading manure is a required field operation if the farmer has livestock. Similarly, municipal sewage treatment facilities often pay their own employees to haul and apply sludge to farmers' fields.

**This 1% decrease in yield is the only decrease attributed to raising the corn organically since we assumed adequate nutrient levels and weed control. In an actual field study, a yield difference of 8.5% was determined between matching plots of organic corn and conventional corn with conventional corn having the higher yield (Lockeretz et al., 1980). This difference was not significant and could be attributed to either inadequate nutrients, weed losses, insect losses, or a combination of one or more of these factors. Organic farmers had higher corn yields than conventional farmers under drought or stress conditions.

TABLE I

Energy inputs and outputs per hectare for conventional and organic corn production in Iowa (Pimentel and Burgess, 1980)

Item	Conventional		Organic											
			Livestock manure		Sewage sludge		Following alfalfa		Following soybeans		Following a 1-year sweet clover fallow	
	Quantity	kcal ha^{-1}	Quantity	kcal ha^{-1}	Quantity	kcal ha^{-1}	Quantity	kcal ha^{-1}	Quantity	kcal ha^{-1}	Quantity	kcal ha^{-1}
Labor (h)	9.6	—	14.8	—	20.3	—	12.2	—	13.6	—	16.6	—
Machinery (kg)	55	990 000	55	990 000	55	990 000	55	990 000	55	990 000	55	990 000
Gasoline (l)	96.7	977 540	96.7	977 540	96.7	977 540	96.7	977 540	96.7	977 540	96.7	977 540
Diesel (l)	26.7	304 526	26.7	304 526	26.7	304 526	26.7	304 526	26.7	304 526	26.7	304 526
LP gas (l)	21.5	165 889	21.5	165 889	21.5	165 889	21.5	165 889	21.5	165 889	21.5	165 889
Electricity (kwh)	27.9	79 906	27.9	79 906	27.9	79 906	27.9	79 906	27.9	79 906	27.9	79 906
Nitrogen (kg)	140.2	1 682 040	25 000 [a]	325 000	46 700 [d]	700 500	28.1[e]	75 000	17 000 [j]	255 000	168 [m]	1387.600
Phosphorus (kg)	72.9	218 670	35.4[b]	46 020	30.4[f]	39 520	67.9[h]	88 270	47.4[k]	61 620	72.9[n]	94 770
Potassium (kg)	84.1	134 560	9.1[c]	20 020	25.7[g]	56 540	69.1[i]	152 020	33.1	72 820	84.1[o]	185 020
Lime (kg)	632.5	199 513	632.5	199 513	632.5	199 513	632.5	199 513	632.5	199 513	632.5	199 513
Seeds (kg)	18.5	462 500	18.5	462 500	18.5	462 500	18.5	462 500	18.5	462 500	18.5	462 500
Insecticides (kg)	2.2	186 857	0	0	0	0	0	0	0	0	0	0
Herbicides (kg)	7.9	792 286	0	0	0	0	0	0	0	0	0	0
Extra cultivations (2×1)	0	0	8	91 300	8	91 300	8	91 300	8	91 300	8	91 300
Transportation (kg)	181.4	46 622	181.4	46 622	181.4	46 622	181.4	46 622	181.4	46 622	181.4	46 622
Total		6 240 909		3 758 836		4 114 356		3 633 086		3 707 236		4 297 586
Corn yield (kg)	8 005	27 884 830	7 925	27 606 156	7 925	27 606 156	7 925	27 606 156	7 925	27 606 156	7 925	27 606 156
kcal output/kcal input		4.47		7.34		6.71		7.60		7.45		5.75
kg output/labor hour (kg h^{-1})		834		5.35		390		650		583		477

[a] 25 wet tonnes of cattle manure applied with an energy input of 15 000 kcal/wet tonne and a labor input of 3.2 h for spreading.
[b] 37.5 kg of P supplied by livestock manure and remaining 35.4 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[c] 7.5 kg of K supplied by livestock manure and remaining 9.1 kg supplied by glauconite (2200 kcal kg^{-1}).
[d] 46.7 wet tonnes (10% solids) of digested sewage sludge applied with an energy input of 15 000 kcal/wet tonne and a labor input of 9.7 h for hauling and spreading sewage sludge.
[e] 42.5 kg of P was supplied by the sludge and remaining 30.4 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[f] 38.4 kg of K was supplied by the sludge and remaining 25.7 kg supplied by glauconite (2200 kcal kg^{-1}).
[g] Nitrogen carryover from alfalfa was calculated to be 112.1 kg ha^{-1} (26), the remaining 28.1 kg of N supplied by 5 t of wet livestock manure with an energy input of 15 000 kcal/wet tonne and a labor input of 0.6 h for spreading.
[h] 7.5 kg supplied by 5 t of livestock manure; 67.9 kg of P supplied by rock phosphate (1300 kcal kg^{-1}).
[i] 15 kg of K supplied by 5 t of livestock manure, 69.1 kg of K supplied by glauconite (2200 kcal kg^{-1}).
[j] Nitrogen carryover from soybeans was calculated to be 44.8 kg ha^{-1} (26); the remaining 95.4 kg was supplied by 17 wet tonnes of manure with an energy input of 15 000 kcal/wet tonne and a labor input of 2.0 h for spreading.
[k] 25.5 kg of P supplied by livestock manure and 47.4 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[l] 51 kg of K supplied by livestock manure and 33.1 kg by glauconite (2200 kcal kg^{-1}).
[m] About 1 200 000 kcal and 6 h of labor are required to plow, plant, mow, and plow-under the sweet clover crop that will provide 168 kg of N.
[n] 72.9 kg of P supplied by rock phosphate at an energy input of 1300 kcal kg^{-1}.
[o] 84.1 kg of K supplied by glauconite at an energy input of 2200 kcal kg^{-1}.

If sewage sludge uncontaminated with heavy metals is substituted for livestock manure, about 46.7 t of sludge is required, or nearly twice as much as livestock manure (Table I). With sewage sludge, rock phosphate, and glauconite providing adequate nutrients, the yield was 7925 kg ha^{-1} with an energy value of 27.6 × 10^6 kcal and a labor input of 20.3 h, giving an energy production ratio of 6.71, or 50% better than conventionally grown corn. However, the labor productivity is only 390 kg h^{-1} or 53% poorer than conventional corn. Three rotations of corn were considered: corn after alfalfa; corn after soybeans; and corn after sweet clover. In each rotation, energy inputs were less than conventional farming inputs. If corn follows a mature stand of alfalfa that was cut for harvest the preceding years and is plowed under in the spring prior to corn planting, the alfalfa provides about 112 kg of N (Table I). The yield is again calculated to be 7925 kg with a labor input of 12.2 h, and the energy production ratio is 7.60, or 70% better than conventionally grown corn. If corn is planted after soybeans, the nitrogen carryover from soybeans is 44.8 kg. The energy production ratio is 7.45 or 67% better than conventionally grown corn. However, the yield of corn per hour of labor is 583 kg, or 30% poorer than conventional corn production (Table I).

If corn is planted after a 1-year fallow planting of sweet clover, the plowed-down sweet clover provides about 168 kg of N (Table I). Sweet clover—corn rotation is assumed to have been practiced for at least 10 years, thus the mineralization of N from the clover crop has reached an equilibrium in N availability. The corn yield is again calculated to be 7925 kg with a labor input of 16.6 h. Thus, the kcal output in corn per kcal energy input is 5.75, or 29% better than conventional corn production. However, the yield of organically grown corn compared with hours of labor is only 477, or 43% poorer than conventionally grown corn. Also, it should be noted that if the yield was calculated on a per hectare basis, then the corn yield would be less than half that of conventional corn production because 2 ha of land (one of corn and one of sweet clover) had to be used to produce the 7925 kg of corn (Table I).

Overall, organic corn production was 29—70% more energy efficient than conventional production. It was most efficient when corn was planted after alfalfa in crop rotation. All organic agricultural systems required a larger labor input than conventional corn systems and the organic labor productivity ranged from 22 to 43% lower than conventional systems.

Wheat

For this analysis of wheat produced under organic and conventional agricultural systems, energy, labor, and yield data were used for wheat grown in North Dakota (Briggle, 1980). Input data for conventional wheat production are listed in Table II. For organic wheat production the synthetic fertilizers were replaced with either livestock manure or sewage sludge (Table II).

TABLE II

Energy inputs and outputs per hectare for conventional and organic spring wheat production following crops in North Dakota (Briggle, 1980)

	Conventional		Organic			
			Livestock manure		Sewage sludge	
Item	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}
Labor (h)	4.5	—	5.8	—	8.4	--
Machinery (kg)	19.0	342 360	19.0	342 360	19.0	342 360
Gasoline (l)	26.6	268 798	26.6	268 798	26.6	268 798
Diesel (l)	46.3	528 925	46.3	528 925	46.3	528 925
Electricity (kwh)	13.3	38 192	13.3	38 192	13.3	38 192
Nitrogen (kg)	67.3	807 360	12 000 [a]	180 000	22 433 [d]	336 000
Phosphorus (kg)	25.8	77 370	7.8 [b]	10 140	25.0 [e]	0
Potassium (kg)	7.0	11 120	0 [c]	0	0 [f]	0
Seeds (kg)	104.3	312 870	104.3	312 870	104.3	312 870
Insecticides (kg)	0.3	26 073	0	0	0	0
Herbicides (kg)	1.7	173 843	0	0	0	0
Transport (kg)	182.6	46 938	182.6	46 938	182.6	46 938
Total		2 633 849		1 728 223		1 874 083
Wheat yield (kg)	1 900	6 279 500	1 824	6 028 320	1 824	6 028 320
kcal output/kcal input		2.38		3.49		3.22
kg output/labor hour (kg h^{-1})		422		314		217

[a] 12 wet tonnes of cattle manure applied with an energy input of 15 000 kcal/wet tonne and a labor input of 1.3 h for spreading.
[b] 18 kg of P supplied by livestock manure and remaining 7.8 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[c] all 7.0 kg of K supplied by livestock manure.
[d] 22.4 wet tonnes of sewage sludge applied with energy input of 5000 kcal/wet tonne and a labor input of 3.9 h for hauling and spreading.
[e] approximately 25 kg of P supplied by sewage sludge.
[f] all 7.0 kg of K supplied by sewage sludge.

The yield of wheat grown by conventional agricultural technology was 1900 kg ha^{-1} with a food energy value of 6.3 × 10^6 kcal and labor input of 4.5 h (Table II). Thus, the energy production ratio of wheat was 2.38 with a labor productivity of 422 kg h^{-1}.

The organic production of wheat used no insecticides, fungicides, or herbicides. Host plant resistance and desired planting times were employed for both insect and disease control. These technologies limited additional insect and plant pathogen losses to 2 and 0%, respectively, with organic wheat production (Pimentel et al., 1979). Substituting good mechanical tillage practices for herbicides in weed control, a 2% increase in weed losses was assumed (Pimentel et al., 1979). Therefore, a 4% reduction in wheat yield was assumed due to insects and weeds. Raising wheat organically using cattle manure is calculated to produce only about 1824 kg ha^{-1} with a food energy value of 6.0 × 10^6 kcal and a labor input of 5.8 h (Table II). The kcal out-

put in wheat per kcal of fossil energy input or energy production ratio is 3.49, or 47% better than conventional wheat production. However, the yield of wheat per labor hour was 314 kg, or 26% lower than the conventional wheat system.

If sewage sludge is substitued for livestock manure, about 22.4 t of sludge is required (Table II). With sewage sludge providing all the N and some P and K, the yield was 1824 kg ha^{-1} and the energy production ratio was 3.22, or 35% better than the conventional wheat production. However, the yield of wheat per labor hour was only 217 kg, or 49% poorer than conventional wheat production.

Thus, the organic systems were 35—47% (or one-third) more energy efficient than the conventional wheat production system (Table II). The organic systems, however, averaged 26—49% lower in labor productivity.

Potatoes

For the analysis of potatoes produced under organic and conventional agricultural systems, energy, labor, and yield data were used for potatoes grown in New York State (Schreiner and Nafus, 1980). Data for conventional potato production are listed in Table III. For organic potato production the synthetic fertilizers were replaced by either livestock manure, digested sewage sludge, or a sweet clover fallow (Table III).

The yield of potatoes grown employing conventional agricultural technology was 32 000 kg ha^{-1} with an energy value of 20.3×10^6 kcal and a labor input of 35 h (Table III), making the energy production ratio 1.28 and the labor productivity 943 kg h^{-1}.

The organic production of potatoes in this example used no synthetic insecticides, fungicides or herbicides (Table III). For alternative weed control, it was assumed that five additional tractor cultivations were made requiring inputs of five additional labor hours and 50 l ha^{-1} of diesel fuel.

No effective non-chemical methods are available for the control of insects and diseases in potatoes; thus insect losses were estimated to increase 20% and disease losses 30% without the use of pesticides (Oelhaf, 1978; Pimentel et al., 1979). Although these losses are not necessarily additive, we estimated total yield losses at 50%. The 50% loss figure, however, does agree with the data of Oelhaf (1978). Thus, total yields were reduced by one-half compared with conventional potato production (Table III).

Raising potatoes organically using cattle manure, therefore, is calculated to produce only 16 500 kg ha^{-1} with a food energy value of 10.1×10^6 kcal and a labor input of 45 h (Table III). The energy production ratio for potatoes grown organically was 1.20, or 7% poorer than conventional potato production. In addition, the yield of potatoes per labor hour was only 367 kg or 61% poorer than the conventional system.

If sewage sludge is substituted for livestock manure, about 76 t of sludge is required to provide adequate nitrogen (Table III). With sewage sludge pro-

TABLE III

Energy inputs and outputs per hectare for conventional and organic potato production in New York (Schreiner and Nafus, 1980)

Item	Conventional		Organic (livestock manure)		Organic (sewage sludge)		Organic (following one year of sweet clover fallow)	
	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}
Labor (h)	35	—	45	—	56	—	47.5	—
Machinery (kg)	14	252 000	14	252 000	14	252 000	14	252 000
Gasoline (l)	261	2 638 449	261	2 638 449	261	2 638 449	261	2 638 449
Diesel (l)	152	1 734 928	152	1 734 928	152	1 734 928	152	1 734 928
Electricity (kwh)	45.7	130 839	45.7	130 839	45.7	130 839	45.7	130 839
Nitrogen (kg)	229	2 748 000	40 820[a]	612 300	76 000[d]	1 140 000	229[g]	1 365 000
Phosphorus (kg)	390	1 170 000	327[b]	425 100	306[e]	397 800	375[h]	487 500
Potassium (kg)	222	355 200	100[c]	220 100	161[f]	354 200	190[i]	418 000
Seed (kg)	2 134	1 309 347	2 134	1 309 347	2 134	1 309 347	2 134	1 309 347
Insecticides (kg)	31.4	2 678 420	0	0	0	0	0	0
Herbicides (kg)	18.0	1 798 380	0	0	0	0	0	0
Fungicides (kg)	6	390 000	0	0	0	0	0	0
Weed control (diesel) (l)	0	0	50	465 000	50	465 000	50	465 000
Transportation (kg)	2 473	635 561	2 473	635 561	2 473	635 561	2 473	635 561
Total		15 841 124		8 423 624		9 058 124		9 436 624
Potato yield (kg)	33 000	20 262 000	16 500	10 131 000	16 500	10 131 000	16 500	10 131 000
kcal output/kcal input		1.28		1.20		1.12		1.07
kg output/labor hour (kg h^{-1})		943		367		295		347 500

[a] 40.8 wet tonnes of cattle manure applied with an energy input of 15 000 kcal t^{-1} and a labor input of 5 h for spreading.
[b] 63 kg of P supplied by livestock manure and remaining 327 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[c] 122 kg of K supplied by livestock manure and remaining 100 kg supplied by glauconite (2200 kcal kg^{-1}).
[d] 76 wet tonnes of sewage sludge applied with an energy input of 15 000 kcal t^{-1} and a labor input of 16 h for hauling and spreading.
[e] 84 kg of P supplied by sludge and remaining 306 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[f] 61 kg of K supplied by sludge and remaining 161 kg supplied by glauconite (2200 kcal kg^{-1}).
[g] about 1 200 000 kcal and 6 h of labor are required to plow plant, mow, and plow-under the sweet clover crop that will provide 168 kg of N. Eleven tonnes (wet) of cattle manure was applied with an energy input of 15 000 kcal t^{-1} to add an additional 61 kg of N. To spread the manure 1.5 h of labor were needed.
[h] 15 kg of P supplied by livestock manure and remaining 375 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[i] 32 kg of K provided from manure and remaining 190 kg supplied by glauconite (2200 kcal kg^{-1}).

viding all of the N and some P and K, the potato yield is 16 500 kg ha^{-1} with a food energy value of 10.1×10^6 kcal. Thus, the energy production ratio is 1.12, or 13% poorer than conventionally grown potatoes, and the labor productivity was only 295 kg h^{-1} or 69% poorer than conventional potato production.

If potatoes are planted after a sweet clover fallow, and the sweet clover is plowed down, this provides about 168 kg of N (Table III). The potato yield is again about 16 500 kg ha^{-1} with a food energy value of 10.1×10^6 kcal and a labor input of 47.5 h. The energy production ratio is only 1.07, or 20% poorer than conventional potato production. In addition, the yield per labor hour was only 347 kg or 63% poorer than conventional potato production. It should also be noted in this sweet clover system that 2 ha of land are required per hectare of potatoes grown.

In conclusion, due to increased insect and disease losses in organic potato production, both energy efficiency and yield per labor hour were substantially lower than conventional potato production (Table III).

Apples

Apple production using conventional and organic agricultural technologies are also compared using apple production data for the northeast (Funt, 1980). The yield for conventional apple production is about 42 t ha^{-1} with an input of 26.1×10^6 kcal and a labor input of 176 h (Table IV). The energy production ratio is 0.89 with a yield of 236 kg of apples per labor hour (Table IV).

If no pesticides are employed in the organic apple system and current apple quality standards are used, we assume that apple yield would be only about 5% that of the conventional system (Pimentel et al., 1979). The apple losses are due to insect pests and plant pathogens (Pimentel et al., 1979). In this case, we assumed that there are no effective alternative insect and pathogen controls to reduce pest losses below 95%; alternative weed controls are employed (Table IV). The weed control substitute for herbicides consists primarily of mowing. The total input for this non-chemical weed control is 3 h of additional labor plus 30 l ha^{-1} of fuel (Table IV).

Therefore, raising apples organically using cattle manure is calculated to produce only 2077 kg ha^{-1} with a food energy value of 1.2×10^6 kcal and a labor input of 180 h (Table IV). The number of labor hours remained approximately the same since apples still require picking and sorting. The energy production ratio is 0.06, or 93% poorer than conventional apple production (Table IV). In addition, the yield of apples per labor hour is only 12 kg or 95% poorer than conventional apple production.

If the apple orchard is seeded with red clover to provide the essential nitrogen for the trees, then an input of about 1.5×10^6 kcal is required for the red clover (Table IV). The apple yield is again about 2077 kg ha^{-1} with a food energy value of 1.2×10^6 kcal but a labor input of 188 h. Thus, the

TABLE IV

Energy inputs and outputs per hectare for conventional and organic apple production in the Northeast (Funt, 1980)

	Conventional		Organic			
			livestock manure		red clover planting	
	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}	Quantity ha^{-1}	kcal ha^{-1}
Labor (h)	176	—	180	—	188	—
Machinery (kg)	82	898 100	82	898 100	82	898 100
Gasoline (l)	1 101	11 130 009	1 101	11 130 009	1 101	11 130 089
Diesel (l)	439	5 010 746	439	5 010 746	439	5 010 746
Electricity (kwh)	20	57 260	20	57 260	20	57 260
Nitrogen (kg)	82	1 205 400	14 700[a]	220 500	168[d]	1 500 000
Phosphorus (kg)	114	627 000	92[b]	119 600	114[e]	148 200
Potassium (kg)	114	231 408	70[c]	154 000	114[f]	250 800
Lime (kg)	682	1 437 656	682	1 437 656	682	1 437 656
Insecticide (kg)	47	2 889 090	0	0	0	0
Fungicide (kg)	49	1 360 730	0	0	0	0
Herbicide (kg)	6	599 460	0	0	0	0
Weed control (diesel) (l)	0	0	30	342 420	0	0
Transportation (kg)	2 716	698 012	2 716	698 012	2 716	698 012
Building (450 sq. ft)	—	14 250	—	14 250	—	14 250
Total		26 159 121		20 082 553		21 145 113
Apple yield (kg)	41 546	23 265 760	2 077	1 163 300	2 077	1 163 300
kcal output/ kcal input		0.89		0.06		0.06
kg output/labor hour (kg h^{-1})		236		12		11

[a] 14.7 wet tonnes of cattle manure applied with an energy input of 15 000 kcal t^{-1} and a labor input of 1.5 h for manure spreading.
[b] 22 kg of P supplied by livestock manure and remaining 92 kg supplied by rock phosphate (1300 kcal kg^{-1}).
[c] 44 kg of K supplied by livestock manure and remaining 70 kg supplied by glauconite (2200 kcal kg^{-1}).
[d] about 1 500 000 kcal and 12 h of additional labor are required to plant the red clover crop in the orchard that will provide 168 kg of N to the apple trees.
[e] 114 kg of rock phosphate at 1300 kcal kg^{-1}.
[f] 114 kg of glauconite at 2200 kcal kg^{-1}.

energy production ratio is only 0.06, or 95% poorer than conventional apple production and the yield per labor hour is only 11 kg. Hence, for both food energy yield and labor productivity, the organic system is about 95% poorer than conventional apple production (Table IV). If 'cosmetic standards' lower than current wholesale/retail standards were acceptable for apples grown organically, then higher apple yields and improved energy input/output ratios would be possible.

SUMMARY

A comparison was made of the fossil energy, labor, and extra land inputs for the production of corn, wheat, potatoes, and apples employing organic technologies (without synthetic chemical fertilizers and pesticides) and conventional agricultural technologies. Nutrients in the organic system were supplied by either livestock manure, sewage sludge, legumes, rock phosphate, or glauconite. Herbicides were replaced by mechanical cultivation and mowing. Except for the use of crop rotations in corn, and host plant resistance in wheat, non-chemical controls for most insect pests and plant pathogens were assumed to be unavailable. Hence, crop yields were reduced for losses due to pests in the organic systems using published crop-yield data.

The results suggested that organic corn and wheat production was 29—70% more energy efficient than conventional production. However, in terms of labor, corn and wheat produced with organic technologies showed 22—53% lower labor productivity.

In contrast, potatoes and apples were less energy efficient (10—90%) to produce organically than by conventional technology. When these crops were grown without pesticides, insect pest and disease losses increased. Labor productivity for organically grown potatoes and apples was 61—95% less than conventional production.

Organic agricultural systems often employ "best management practices" (USDA, 1980) that include sod-based rotations, cover crops, and green manure crops. Indirect benefits of these organic technologies can be: reduced soil erosion; reduced water runoff rates and conservation of water; and increased organic matter in the soil and associated beneficial soil biota. The disadvantages of organic agriculture can be increased weed problems (e.g., weed seeds in manure) and reduced soil moisture resulting from growing a legume for nitrogen.

Although this analysis suggested that organic corn and wheat production was more energy efficient than conventional production, the adoption of organic technologies has several constraints. First, labor productivity averaged 22—95% less than for conventional production. Our analysis may have exaggerated some labor costs by including labor input for manure hauling and spreading; there is no doubt, however, that labor inputs are substantially greater for organic technology. Lockeretz et al. (1981) calculated an increase of 12% per unit value of crop produced organically compared with conventional production, and Oelhaf (1978) calculated about a 20% greater labor input for organic crops. All of these studies used different methods of assessing labor inputs, and therefore are not fully comparable. Historically, American farmers have substituted capital for labor and this trend is continuing (Buttel and Gertler, 1982).

Another constraint is the availability of adequate quantities of organic fertilizer like manure (USDA, 1980). For example, only about half of the farms now in Iowa keep cattle that would provide a source of manure (USBC, 1981). This reflects the growing tendency for specialization in U.S. agriculture (Fast and Gertler, 1981).

A major limitation of this study rests on the use of energy, crop yield, and labor data from unrelated studies. Clearly, sound field studies of both organic and conventional agricultural technologies are needed for corn, wheat, potatoes, apples, and other major crops.

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Agriculture, Ecosystems and Environment, 40 (1992) 3–23
Elsevier Science Publishers B.V., Amsterdam

Agroecosystem biodiversity: matching production and conservation biology

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ABSTRACT

Paoletti, M.G., Pimentel, D., Stinner, B.R. and Stinner, D., 1992. Agroecosystem biodiversity: matching production and conservation biology. *Agric. Ecosystems Environ.*, 40: 3–23.

A review of the existing literature on biodiversity connected with agricultural activities has been developed, and the possible sustainable alternatives have been looked into.

Following recent evaluations, only one-twentieth to one-sixtieth of the planet's species have yet been described and most of these will be lost if the destruction of the environment continues at its present rate. Most of the terrestrial environment (up to 95%) is affected by human activities including agriculture and the terrestrial habitats provide up to 98% of human food on the planet. Sustainable strategies in food production in agriculture improve the existing biodiversity and include the following items: increased porosity of the landscape through proper management of natural vegetation, better use and recycling of organic residues, introduction of integrated farming systems, reduced tillage, rotation, biological control, increased number of biota involved in human food-webs.

INTRODUCTION

Why are we concerned about the biotic diversity in space, in time in the anthropized cultivated areas? Is the sufficient production of foods, fiber and meat reasonably consistent with the conservation of biodiversity? Biodiversity is associated with terrestrial areas in which diversity of natural biota play the same social and economic roles as do "libraries, universities, museums, symphony halls, and newspapers. They must be integrated with the educational system in the same way as these other complex information storage and transfer systems" (Janzen, 1988). Most demographic, social, economic and environmental problems occur in tropical areas and it is difficult to convince the public about the 'library excellence' of natural biota when they are with-

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TABLE I

Numbers of described species of living organisms (from Wilson, 1988)

Kingdom and major subdivision	Common name	No. of described species	Totals
Virus	Viruses	1000 (order of magnitude only)	1000
Monera			
Bacteria	Bacteria	3000	
Myxoplasma	Bacteria	60	
Cyanophycota	Blue–green algae	1700	4760
Fungi			
Zygomycota	Zygomycete fungi	665	
Ascomycota (including 18000 lichen fungi)	Cup fungi	28650	
Basidiomycota	Basidiomycete fungi	16000	
Oomycota	Water molds	580	
Chytridiomycota	Chytrids	575	
Acrasiomycota	Cellular slime molds	13	
Myxomycota	Plasmodial slime molds	500	46983
Algae			
Chlorophyta	Green algae	7000	
Phaeophyta	Brown algae	1500	
Rhodophyta	Red algae	4000	
Chrysophyta	Chrysophyte algae	12500	
Pyrrophyta	Dinoflagellates	1100	
Euglenophyta	Euglenoids	800	26900
Plantae			
Bryophyta	Mosses, liverworts, hornworts	16600	
Psilophyta	Psilopsids	9	
Lycopodiophyta	Lycophytes	1275	
Equisetophyta	Horsetails	15	
Filicophyta	Ferns	10000	
Gymnosperma	Gymnosperms	529	
Dicotyledoneae	Dicots	170000	
Monocotyledoneae	Monocots	50000	248428
Protozoa	Protozoans: Sarcomastigophorans, ciliates, and smaller groups	30800	30800
Animalia			
Porifera	Sponges	5000	
Cnidaria, Ctenophora	Jellyfish, corals, comb jellies	9000	

Kingdom and major subdivision	Common name	No. of described species	Totals
Platyhelminthes	Flatworms	12200	
Nematoda	Nematodes (roundworms)	12000	
Annelida	Annelids (earthworms and relatives)	12000	
Mollusca	Mollusks	50000	
Echinodermata	Echinoderms (starfish and relatives)	6100	
Arthropoda	Arthropods		
Insecta	Insects	751000	
Other arthropods		123161	
Minor invertebrate phyla		9300	989761
Chordata			
Tunicata	Tunicates	1250	
Cephalochordata	Acorn worms	23	
Vertebrata	Vertebrates		
Agnatha	Lampreys and other jawless fishes	63	
Chrondrichthyes	Sharks and other cartilaginous fishes	843	
Osteichthyes	Bony fishes	18150	
Amphibia	Amphibians	4184	
Reptilia	Reptiles	6300	
Aves	Birds	9040	
Mammalia	Mammals	4000	43853
Total — all organisms			1392485

out enough food for their families and they live in absolute poverty (De Miranda and Mattos, 1992).

Organic material equivalent to 38.8% of the present net primary production of the planet is today appropriated entirely by humans (Vitousek et al., 1986). One-third of the global land surface of the planet still is wilderness (48.069×10^6 km^2) (McCroskey and Spalding, 1989) but most of this area has humans living in the wilderness regions (Western, 1989).

The amount of species on the planet (that means biological units, which are diverse genetically and ecologically) is estimated between 30 and 50 million according to Erwin (1982, 1988). Only 1 392 485 (Table 1) species are scientifically described according to some inventories (Parker, 1982; Arnett, 1985; Kevan, 1985; Wilson, 1988; Minelli, 1989; Pimentel et al., 1992a) or something more, 1.7–1.8 million (Stork, 1988; May, 1990). Discrepancies between the amount forecasted and the described number of species implies defects in understanding our planet's structure, composition, diversity and

complexity. No matter that the number of biota in the planet is far larger than, for instance, the artefacts. The number of chemical compounds artificially created by our technology is difficult to quote but 60 000 chemicals are presently in use in the US (National Toxicology Program, 1981). However the US is home for an estimated 500 000 living species of which 95% are small organisms (Knutson, 1989).

An inventory of the warehouse is needed before we can elaborate the strategy for the use or manufacture of goods. On our planet, knowledge of biota is limited and scanty. For instance, there are only 3000 scientists who specialize in tropical biology and 1500 taxonomists who are devoted to the tropical areas (Raven, 1980; Myers, 1985). Can we use the existing biodiversity better and can we conserve it? Can we assign a real value to biotic diversity? It has been argued: "I cannot help thinking that when we finish assigning values to biological diversity we will find that we do not have very much biological diversity left" (Erenfeld, 1988).

On a global scale of ecology and brain-storming the Gaia Hypothesis related to the interactions of biotic activity and most climatic and abiotic factors (Lovelock, 1989; Hinkle and Margulis, 1990), we need efforts to investigate the composition and differentiation of the living biota. In particular, interactions between agriculture and biodiversity are important (Pimentel et al., 1992a). About 70% of the earth's ecosystems (50%, agriculture including pastures; 20%, forests) are manipulated to obtain 98% of the food fiber and wood used by humans (Food and Agriculture Organization, 1981; Vitousek et al., 1986; Western, 1989; Pimentel et al., 1992a). In addition, about 25% of the land is devoted to urbanization and human settlements. Thus, humans occupy and manage approximately 95% of the earth's terrestrial ecosystems (Western, 1989).

We maintain that each species has intrinsic value. Earthworms, insects, herbs, fungi or bacteria are no less fascinating or less worthy of our interest and conservation than the larger 'Noah's Ark' organisms such as elephants, birds and sequoias (Pimentel et al., 1992a).

Some questions that should be addressed are: which part of the existing biota is living in cultivated or human occupied areas which cover the majority of the terrestrial areas; how many are living in wild protected areas; are the two components interchangeable; are the biota living in the wilderness possible pests for crops; can the diversity of biota be used as a tool for development and improve the well-being of humans?

We are not capable of giving exhaustive responses to these questions but we hope to demonstrate that opportunities exist to conserve biodiversity in agriculture and forestry.

THEORETICAL BASES OF BIODIVERSITY ESTIMATIONS

At least three guidelines provide estimates of the biotic diversity on the planet. These include the following.

Food-web structures and dimensions

The richness of a food-web and the length of food-chains are predicted to be higher in tropical areas than in temperate habitats (Pimm and Lawton, 1977; Kitching and Pimm, 1985; Kitching, 1990). Because of food-web complexity and number, food-web abundance implies a large number of species. In tropical areas, food-webs are more numerous than in temperate areas (Pimm and Lawton, 1977; Cohen et al., 1990). In cultivated areas many food-webs are missing or unstable (Odum, 1984). For instance, pesticide residues affect soil detritivores and predators (Pimentel and Edwards, 1982; Paoletti et al., 1988, 1992; Pimentel et al., 1991).

Body dimensions

Data are projected from some regression figures obtained by plotting the mean body length and the number of species (May, 1986). Using the body length assumption for small organisms (0.5 mm) to be similar for all organisms the rate of known and unknown species is one to 10, that means 10 million species on the planet (May, 1986, 1988, 1990).

Extrapolations following geographical endemism and host specificity

Based on an examination of the geographic and host specificity, May (1988, 1990) stated that an estimate of animal species is possible. Based on the estimated 300 000 living plant species in the tropics and 10 insect species per tree, it is calculated that there are 3 000 000 species of associated insects.

Erwin (1982) calculated that with 600 non-shared beetle species per tree and 50 000 tree species, the total beetle species is 30 million. More recently Erwin (1988) sorting the materials collected in the canopy of the Tambapora rain forest in Peru has estimated the value to be more than 50 million insect species. Based on this estimation and current deforestation, Erwin reported that several million species will be lost if forest destruction is not stopped. These estimates were discussed by Stork (1988), Adis (1990), May (1990), Hodkinson and Casson (1991) and Paoletti et al. (1990, 1991a). In particular Stork (1988) estimates between 7.3 and 81.4 million arthropod species in the tropical rain forests. Other estimations based on species–area curves have been given (Simberloff, 1986; Reid and Miller, 1989).

DESTRUCTION RATE AND LOSS OF INFORMATION

From 250 000 to 750 000 plant species are estimated to exist in the world, with 65% found in the tropics (Farnsworth, 1988; Wilson, 1988). Chemical, folklore and pharmaceutical data are found for as many as 25 000 plant spe-

TABLE 2

Estimates of potential species extinction in the tropics

Estimate	Basis of estimate	Source
From 1950 to 1980 one half of previously living species		Whittaker and Likens, 1975
One species day^{-1} to one species hr^{-1} between 1970s and 2000	Unknown	Myers, 1979
33–50% of all species between 1970s and 2000	A concave relationship between percentage of forest area loss and % of species loss	Lovejoy, 1980
A million species or more by end of this century	If present land-use trends continue	National Research Council, 1980
As high as 20% of all species	Unknown	Lovejoy, 1981
50% of species by the year 2000 or by the beginning of next century	Different assumptions and an exponential function	Ehrlich and Ehrlich, 1981
Several hundred thousand species in just a few decades	Unknown	Myers, 1982
25–30% of all species, or from 500000 to several million by the end of this century	Unknown	Myers, 1983
500000–600000 species by the end of this century	Unknown	Oldfield, 1984
33% or more of all species in the 21st century	Present rates of forest loss will continue	Simberloff, 1983
20–25% of existing species by the next quarter of century	Present trends will continue	Norton, 1986
15% of all plant species and 2% of all plant families by the end of this century	Forest regression will proceed as predicted until 2000 and then stop completely	Simberloff, 1986
0.75 million species by the end of this century	All tropical forests will disappear and half their species will become extinct	Raven Missouri Botanical Gardens, (Lugo, 1988)
20–30 million species	If 50% of rainforests are destroyed by the year 2000	Erwin, 1988

cies (Napralert data base, Farnsworth, 1988). However only in 119 species have pure chemical substances been extracted from these plants and used in medicine (Farnsworth et al., 1985). Approximately 3500 are new chemical structures reported from natural sources (higher plants, lichens, filamentous fungi, bacteria, animals) (Farnsworth, 1988). Few plants have gained importance as pesticides; this is due to the lack of scientific attention rather than lack of pesticidal activities in plants (Grainge and Saleem, 1988; Yang and Tang, 1988). For instance, alkaloids which are useful compounds derived from flowering plants have been examined only in 5000 plants, 2% of the estimated 250 000 plants (Eisner, 1990).

Some investigators have tried to measure the loss of species associated with the rain forest destruction (Table 2) and have calculated the loss to be approximately five million species. Forest destruction each year causes the extinction of about 17 500 species. This extinction of species is 1000–10 000 times greater than the geological extinction of species (Raup, 1988; Wilson, 1988). This geological extinction rate is estimated based on marine organisms and we know that the species evaluation conducted by paleontologists is different if compared with living biotic taxa present in terrestrial ecosystems. Clearly the rate of species extinction today is significantly higher than it was throughout geological time.

EDIBLE PLANTS AND BIODIVERSITY

Although 150 plants are commonly eaten worldwide, only 15 plant species provide more than 90% of world food (Pimentel and Pimentel, 1979; Plotkin, 1988) and only three (rice, corn and wheat) produce almost two-thirds of this amount (Raven, 1988).

Ninety-eight percent of US crop production is based on species originating outside US boundaries (Caufield, 1982) meaning that most cultivated plants are located outside of the region of origin. A single tribe of Amazonian Indians may use more than 100 different native species of plants for medical purposes alone (Prance, 1982; Plotkin, 1988) and at least 90 tribes have disappeared since the beginning of this century (Plotkin, 1988). A large number of plants (212) have been used by an Amazonian tribe, the Quichuas, in Ecuador (Alarcon Gallegos, 1988). In temperate zones, forests have yielded only about 20 major edible fruits (Myers, 1985).

In New Guinea 251 tree species bear edible fruit and only 43 are cultivated crops (Jong, 1979). In 50 ha of primary lowland forest in Malaysia 820 tree species have been identified and 76 are known to bear edible fruits (Saw et al., 1991). There could well be 2500 fruit species in tropical forests worldwide for human consumption and an estimated 250 of these species are thought to be widespread in the tropics (Myers, 1985).

In northeastern Italy 42 wild herbs are collected in spring to prepare the

dish 'pistic'. However, only two plant species are actually cultivated (Dreon et al., 1992). In the Pacific Yap Islands, farmers cultivate over 50 fruit tree species (Falanruw, 1989); in Indonesia in the farm gardens more than 500 species are cultivated (Michon, 1983). In Swaziland, 220 wild plant species are commonly consumed (Ogle and Grivetti, 1985). Large amounts of plant biomass and diversity indicate more invertebrate and microbic diversity than low amounts of biomass (Pimentel et al., 1992a).

Many examples exist of useful food diversity from tropical regions. For instance, sweet tomato (*Lycopersicon esculentum*) comes from a wild tomato (*Lycopersicon chmielewskii*) which was collected in Peru by Iltis (1988). Many tropical plants provide promising new food sources (Myers, 1985; King and Gershoff, 1987; Prance et al., 1987; Williams, 1988; Altieri and Merrick, 1988; Bohs, 1989; Reid and Miller, 1989; Wilson, 1990; Saw et al., 1991).

Diversity of wild invertebrates, both terrestrial and aquatic, and vertebrates have also been utilized as human food by native tribes in tropical areas (Dufour, 1987; Posey, 1987; Defoliart, 1989; Pimentel et al,. 1992a; Carpaneto and Germi, 1992) and are suggested as an effective food resource (Kov et al., 1988).

Forest relicts in the mosaic agroecosystem

Numbers of plant species are indicators of old woodlands (Peterken, 1974) and the number of invertebrates such as ground beetles and soil invertebrate species seem to accumulate with the increased age of the woodland (Brandmayr, 1983; Paoletti, 1988; Terrell-Niedl, 1990). In the case of ground beetles (Coleoptera) the area of the woodland is not a significant factor in predicting the number of species (Terrell-Niedl, 1990).

Better drained woodland remnants in the Pianura Padana accumulate more soil invertebrate species and biomass than heavily wet forest soils (Paoletti, 1988). Coarse woody debris including standing dead trees increase the species diversity in forested areas (Elton, 1966; Thomas, 1979).

Deciduous woodland remnants affect diversity in the agroecosystem landscape in northeastern Italy. Most predators found in the woodland lots can be found in the surrounding crops (grape, wheat, corn, soy bean and weeds on dikes) (Paoletti et al., 1989; Favretto et al., 1991). In the landscape, hedgerows and woodland remnants can modify the predator and invertebrate body size and overall biodiversity (Karg, 1980, 1989; Ryskowski et al., 1991).

Grasslands

The number of naturally living species can be as large as 250–300 plants in a range of 250 ha within the US Central Plains (Risser, 1988). The number

TABLE 3

Animal biological diversity in various ecosystems

Ecosystem	Location and area	Animal species	Source
Pasture	Britain (ha)	1000	MacFadyen, 1961
Forest, tropical	Borneo (10 trees)	2800[b]	Stork, 1988
Forest, beech	W. Germany (forest)	1500–1800	Elienberg et al., 1986
Forest, tropical	Costa Rica (10800 ha)	13000[b]	Janzen, 1987
Hungarian national parks	Hungary	4433–8843[d]	Mahunka, 1987
Collards	New York (ha)	262	Root, 1973
Alfalfa	New York (ha)	600[a]	Pimentel and Wheeler, 1973
Corn and woodland remnants	NE Italy	450[c]	Paoletti, 1988
Corn agroecosystems and deciduous woodland remains	NE Italy	800–1500[a,c]	Paoletti, 1988
Sustainable farming systems, Lautenbach: Mesostigmatic mites	Germany	60	El Titi and Landes, 1989
Hungarian corn field	Hungary	598[a]	Meszaros, 1984a
Hungarian apple orchard	Hungary	1759[d]	Meszaros, 1984b

[a]Above-ground arthropods.
[b]Insects only.
[c]Soil invertebrates.
[d]All animals.

of edible wild 'pistic' plants collected in spring in the grassland mosaic in Friuli, Italy, decreases as the grasslands are abandoned (Dreon et al., 1992). Most soil invertebrate biodiversity and biomass is found in grasslands (Edwards and Lofty, 1969; Matthey et al., 1990).

In addition to plants, fungi, bacteria, invertebrates account for the bulk of the biomass in pasture ecosystems as well as in woodlands and agroecosys-

TABLE 4

Biomass of various organisms per hectare in a temperate region pasture

Organism	Pasture biomass (kg fresh weight)
Plants	20000
Fungi	4000[a]
Bacteria	3000[a]
Arthropods	1440[a]
Annelids	1320[a]
Protozoa	380[a]
Algae	200
Nematodes	120
Mammals	1.2[b]
Birds	0.3[b]

[a]Richards (1974).
[b]Walters (1985).

tems (Tables 3 and 4). In contrast mammals and birds contribute only 1.2 kg ha^{-1} and 0.3 kg ha^{-1} (fresh) biomass, respectively.

PLANT AND ANIMAL BIOMASS AND BIODIVERSITY

Biological diversity in an ecosystem is related to the amount of living and non-living organic matter present in the ecosystem (Wright, 1983, 1990). Positive correlations between biomass production and species abundance have been recorded (Allee et al., 1949; Odum, 1971, 1978, 1989; Ricklefs, 1979; Zlotin, 1985; Paoletti, 1988; Hendrix et al., 1989; Pimentel et al., 1992a). However this seems to be not always true, as in caves, for instance, in which larger guano deposits imply lower diversity of animal species at least in the tropics (Sbordoni et al., 1977). Krebs (1985) and Colinvaux (1986) have also contested the universality of this contention. In oligotrophic rain forests, however, the biodiversity is incredibly high (Jordan and Herrera, 1981; Erwin, 1982; Jordan, 1985; Stork, 1988; Wilson, 1988) but in suspended soils, inside bromeliads in the canopy of the tropical rain forests more diversity and more invertebrates have been found compared with the ground soils underneath (Paoletti et al., 1990, 1991a).

In any case, manure introduced in experimental field plots has increased both the collard biomass and animal diversity (Pimentel and Warneke, 1989) (Table 5). In grassland plots in Japan the species diversity of the macrofauna more than doubled when 30 t ha^{-1} (wet) manure was added in the land (Kitazawa and Kitazawa, 1980). Similar data showing an increase of species diversity have been given by Edwards (1983) in Britain, Curry (1987) in Ire-

TABLE 5

Density or biomass increase of soil-dwelling invertebrates or microbes as ratio of control in response to added manure biomass in the UK and Switzerland

Ecosystem	Arthropod increase	Microbe increase	Source
	Earthworms		
Wheat	Two-fold	–	Morris, 1922
Mangolds	Four-fold	–	Morris, 1927
Mangolds	Seven-fold	–	Raw, 1967
Cropland	–	Ten-fold	Olah-Zsupos and Helmeczi, 1987
Cereals	11.6-fold		Matthey et al., 1990

land, Bohac and Pokarzhevsky (1987) in USSR, and Matthey et al. (1990) in Switzerland.

In addition, in forests, grasslands, agroecosystems and hedgerows biological diversity is related to the age of the system (Muir and Muir, 1987; Paoletti, 1988). Hedgerow plant diversity is similar to the woodland deciduous forest (climax situation) in northeastern Italy (Zanaboni and Lorenzoni, 1989) and soil invertebrate diversity (species number) in woodland remnants and corn fields in northeastern Italy seems to be a factor of biomass (Paoletti, 1988). Abandoned agricultural fields accumulate species (Odum, 1960, 1969). The number of herbivores in crops is related to the area covered by the crop culture in a region (Strong et al., 1984; Pimentel et al., 1992a). We estimate there are 800–1500 species of invertebrates living in corn fields in northeastern Italy (Paoletti, 1988); 598 invertebrate species have been reported (mostly on plants) in corn fields in Hungary (Meszaros, 1984a), and 1759 species in apple orchards (Meszaros, 1984b). A corn field in northeastern Italy, may have 200–450 invertebrate species, and a less disturbed lowland deciduous forest could contain 300–500 invertebrate species (Paoletti, 1988). In three national parks in Hungary, 4433, 7384, and 8843 animal species have been listed (Mahunka, 1987). In an northern Italian swamp, 1178 animal species have been recorded (Daccordi and Zanetti, 1989).

AGROECOSYSTEMIC EVOLUTION — STABILITY AND NUMBER OF SPECIES

Plowing reduces soil invertebrate species diversity and also reduces biomass compared with pastures, natural grasslands and alfalfa fields (Kevan, 1962; Edwards, 1983; Lee, 1985; Paoletti, 1988; Paoletti et al., 1988; Dindal, 1989; Stinner and House, 1990). For instance, corn monoculture or grape monoculture reduces detritivores such as earthworms in terms of species number and biomass (Paoletti et al., 1988).

Flying insects like Diptera and Hymenoptera and drifting spiders are more mobile and recover quickly in contaminated (heavy pesticide use) conventional apple orchards but species diversity (Shannon index) is lower than on untreated (biological) orchards. Detritivores and the polyphagous beetles (Carabidae) were depressed in the conventional orchard but were abundant on a biological orchard (Paoletti et al., 1992). Detrivores and predators are affected by meadow management and are increased by the vegetal biomass input (Andrzejewska et al., 1986) and decreased tillage (Stinner and House, 1990).

Margin effects

Margins' vegetal typology differs from crop and wild vegetation borders and this affects the pattern of invertebrates and their abundance (Pimentel, 1961; Taylor et al., 1978; Karg, 1980, 1989; Altieri, 1981; Fauvel and Cotton, 1981; Paoletti, 1984; Paoletti and Lorenzoni, 1989; Paoletti et al., 1989; Dennis and Fry, 1992; Kromp and Steinberger, 1992; Hassall et al., 1992; Lagerlof et al., 1992; Pimentel et al., 1992a). For example, hedgerows affect plant and animal diversity as well as undisturbed field margins. Mosaic structure of the landscape improves the composition and biodiversity of agroecosystems (Noss, 1987; Baudry, 1989; Haber, 1990).

CONCLUDING REMARKS

Landscape structure, field area and margins, as well as polyculture which are part of traditional agriculture appear to increase biodiversity (Altieri et al., 1987; Gliessman, 1990). In some cases there is no clear difference in diversity between the cultivated and natural ecosystem.

For instance, no significant difference exists between the wild vegetation and cultivated crops in some farming situations. For example the Tarahumara Indians in the Mexican sierras use edible weed seedlings (*Amaranthus, Chenopodium, Brassica*) from April through July. This period is right before their maize, bean, squash, chillies and other food crops come into production (Altieri et al., 1987). Traditional home gardens in Costa Rica (Gliessman, 1990) or other tropical areas host a high number of plants and animals as possibly did the Mexican 'Pet Kot' (Gomez-Pompa et al., 1987). Monocultures have reduced vegetal biodiversity and eliminated the vegetal mosaic common in nature. In addition, this reduced the marginal vegetation, including hedgerows and adjoining woodlands, also reducing the amount of organic materials incorporated into the soils (Paoletti and Lorenzoni, 1989; Haber, 1990; Ryskowsky et al., 1991).

The introduction of large amounts of organic matter by recycling crop residues, manure, mulching, reduced tillage practices in fields increase biomass and species numbers (Pimentel and Krumel, 1987; El Titi and Ipach, 1989;

TABLE 6

Farming systems which can increment biodiversity in agroecosystems

Sustained biodiversity	Decreased biodiversity
Hedgerows (1,2)	Wild vegetation removal
Dykes with wild herbage (1,2)	Tubular drainage or vegetation removal
Polyculture (3)	Monoculture
Agroforestry (3)	Monoculture
Rotation with legumes (4)	Monosuccession
Dead mulch, living mulch (4,5)	Bare soil
Strip crops, ribbon cropping (6)	Conventional cropping
Alley cropping (6)	Monoculture
Minimum, no-tillage, ridge tillage (5,7,8)	Conventional plowing
Mosaic landscape structure porosity (9,10,11)	Landscape simplification, woodland clearance
Organic sustainable farming (4,12)	Intensive input farming
On-farm research (13,14)	Conventional plot research
Organic fertilizer (4,12)	Chemical fertilizer
Biological pest control (15)	Conventional chemical pest control
Plant resistance (15)	Plant susceptibility
Germoplasm diversity (3,16,17)	Standardization

(1) Paoletti et al., 1989.
(2) Favretto et al., 1991.
(3) Altieri et al, 1987.
(4) Werner and Dindal, 1990.
(5) Stinner and House, 1990.
(6) Personal evaluations (Paoletti 1987–1990).
(7) Exner et al., 1990.
(8) House and Rosario-Alzgaray, 1989.
(9) Paoletti, 1988.
(10) Noss, 1990.
(11) Karg, 1989.
(12) Matthey et al., 1990.
(13) Stinner et al., 1991.
(14) Lockeretz, 1987.
(15) Pimentel et al., 1991.
(16) Lal, 1989.
(17) Oldfield and Alcom, 1987.

Pimentel and Warneke, 1989; Werner and Dindal, 1990; Matthey et al., 1990). Table 6 is an attempt to incorporate some trends at farm and landscape scales that can be effective in preserving biodiversity and economically sustainable farming systems. Nevertheless, at a regional level appropriate research to compare undisturbed ecosystems and agroecosystems in a historical perspective is also needed. Few data on biodiversity are available for many regions. Clearly, to preserve biodiversity on the planet, increased understanding of managed agricultural and forest ecosystems is needed.

Conserving biodiversity in managed crop and pasture systems provides many opportunities as well as many challenges to scientists and planners. The objectives of sustainable agriculture and biodiversity conservation are economically compatible and should be actively pursued (Altieri et al., 1987; Reid and Miller, 1989; Gliessman, 1990; Edwards et al., 1990; Francis et al., 1990; Paoletti et al., 1991b; Pimentel et al., 1991, 1992a; Stinner et al., 1991; Koehler, 1992).

Some points of desirable features in agroecosystems to promote sustainability and biodiversity comprise the ones mentioned in Table 6.

ACKNOWLEDGMENTS

The authors thank Robert May and Matthew R. Werner for reviewing the manuscript and offering insightful comments for its improvement.

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Water Resources in Food and Energy Production

David Pimentel, Sarah Fast, Wei Liang Chao, Ellen Stuart, Joanne Dintzis, Gail Einbender, William Schlappi, David Andow, and Kathryn Broderick

Conflicts for water between US agriculture and energy sectors are projected to increase. Currently agriculture consumes 83% of the water withdrawn, and an increase of 17% is projected by the year 2000. At the same time, a major synfuel program from coal and oil shale could increase water consumption for energy production 4- to 30-fold over present levels. A significant portion of this growth in water demand will occur in the western states where most irrigated agriculture and one-half of US coal reserves are located. Environmental restrictions and the necessity to maintain aquatic ecosystems are additional constraints. *(Accepted for publication 15 June 1982)*

Water resources are both ecologically and economically essential to the United States. Humans must consume 2 to 3 liters of water per day (l/d) for survival. In addition, to maintain the high standard of living in this country, about 7200 l/d (1900 gal/d) of water per person are withdrawn from rivers, lakes, and underground aquifers. Some of this is used directly by individuals and industry, but almost 72% is used to supply food and energy needs (Murray and Reeves 1977, USWRC 1979). In the United States, the production of the 2 kilograms of food consumed per person each day requires about 2700 l of pumped water plus rainfall. The 30 kg of coal equivalents of energy consumed per capita daily requires pumping an additional 2300 l of water.

Water resources in certain US regions are currently overdrawn, especially in the western United States. Drawdown problems over the Ogallala aquifer in the Great Plains (Boddé 1981, Walsh 1980), water deficits along the Colorado River, and water shortages in the San Joaquin Valley are symptomatic of this pressure (Harrison 1977, USDI 1976). Water shortages are also occurring in certain limited regions of the eastern United States where overall water availability is much higher than in the West (Sheer 1980).

Because both agricultural and energy production processes use large quantities of water, their anticipated growth over the next two decades is expected to tax US water resources as well as many natural ecological systems. Specifically, by the year 2000, food production is projected to increase about 30% to meet domestic and export food needs (ERAB 1981). Projections are that biomass energy production systems will increase from 5- to 20-fold by the year 2000 (ERAB 1981, OTA 1980a) while synfuels from coal and oil from shale might increase from 2- to 5-fold (Livingston et al. 1981, NAS 1979). Indeed, water will be one of the limiting factors of energy production in the future (Harte and El-Gasseir 1978, NAS 1979).

In this report some of the current and future constraints that water shortages may impose on food and energy production are assessed. These include: the availability of usable water in the United States, the current and projected water consumption demands for the agriculture and energy sectors, and the environmental impact of water use on aquatic ecosystems and natural resources. Although many sectors of society require water and may experience water deficits, the analysis is primarily focused on use of water in food and energy production because these are the major consumers of water in the United States.

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WATER AVAILABILITY

An estimated 152,000 billion liters per day (bld) of water vapor drift over the continental United States; about one-tenth of this water precipitates daily (USWRC 1979). Nearly two-thirds of this precipitation is converted back into water vapor by transpiration by plants and evaporation from other surfaces (Garstka 1978, Nace 1976), leaving about 5000 bld of surface and subsurface flow.

The US Water Resources Council (USWRC 1979) estimates that a maximum of 2600 bld of average annual stream flow is available for use. At seasonal low flow periods, only 1350 bld are available nationwide for consumption. The limited available surface storage present in lakes and reservoirs, as well as variable rainfall, make it difficult to increase the available surface supply of water. The construction of additional reservoirs is not practical at present because most of the large, economically and hydrologically feasible reservoir sites have already been developed (Nace 1976).

Water also is available from groundwater; about 10 times more fresh water is stored underground than on the surface in the United States. Estimates of the amount of groundwater range as high as 2.30×10^{17} l, with about 58×10^{12} l of this considered usable (Akerman and Lof 1959).

In general, replenishment of groundwater by rainfall is quite slow. Annually only about 230 bld or <1% of the store of groundwater is replaced (CEQ 1980a). Groundwater that has accumulated over millions of years is being mined today to offset surface and groundwater deficits resulting from population growth and expanded energy and agricultural uses. Thus, withdrawal of groundwater exceeds natural water recharge and recy-

cling. For example, in 1975 about 310 bld of groundwater was pumped (USWRC 1979). In the coterminous states, water overdraft exceeded replenishment by about 80 bld, or 25%, and in the Texas-Gulf area, overdraft is as high as 77%.

CURRENT WATER USE

An estimated 1600 bld of water is withdrawn daily from US water resources, excluding that needed for hydroelectric uses (Murray and Reeves 1977). Of this, about 62% is fresh water withdrawn from surface sources; 20% is from fresh groundwater; 16% from saline surface water; and 2% from saline groundwater. Of the total water withdrawn, the industrial, energy, and agricultural sectors pumped 92%, and public and rural water supplies pumped the remaining 8% (Figure 1).

About 77% of the water withdrawn from surface and subsurface sources is returned to rivers and lakes (Murray and Reeves 1977). The remaining amount or 360 bld of the water that is withdrawn is consumed or lost. Most of this water is lost simply by evaporation and transpiration and is no longer directly available for reuse. Of the estimated 360 bld consumed, only 296 bld can be considered renewable in terms of daily flow based on demand and calculated regional availability of water (Figures 1, 2, and 3). Renewable means that groundwater sources are not reduced and that no more than 40% of the mean annual flow of instream water is depleted so natural aquatic systems are not seriously affected (Tennant 1976).

Agricultural irrigation consumes 83% of the total of 360 bld that is consumed or lost (Figure 1). Most of the water consumed by agriculture is lost by transpiration and evaporation. The public, industrial, and rural sectors consume the remaining 17% of the total (Figures 1 and 3).

On a per capita basis, an estimated 7200 l/d of water is withdrawn for each American (Table 1). Note that the per capita withdrawal in the western region is twice that in the eastern region. The major difference between these regions is in water consumed, with more than 12 times as much fresh water consumed in the West than in the East. The high rate of consumption in the West, primarily for irrigation, accounts for both the water imbalances and the mining of groundwater to offset water deficits (USWRC 1979).

Withdrawals and consumption are expected to increase, especially in western areas (USWRC 1979). The highest projected increases in domestic consumption are for four regions: the Texas Gulf, lower Colorado, Great Basin, and California (Figure 2). Although these regions account for 30% of the projected increase in domestic water consumption, California alone accounts for 20%. This trend in water consumption parallels expected population growth in the sunbelt areas of the United States.

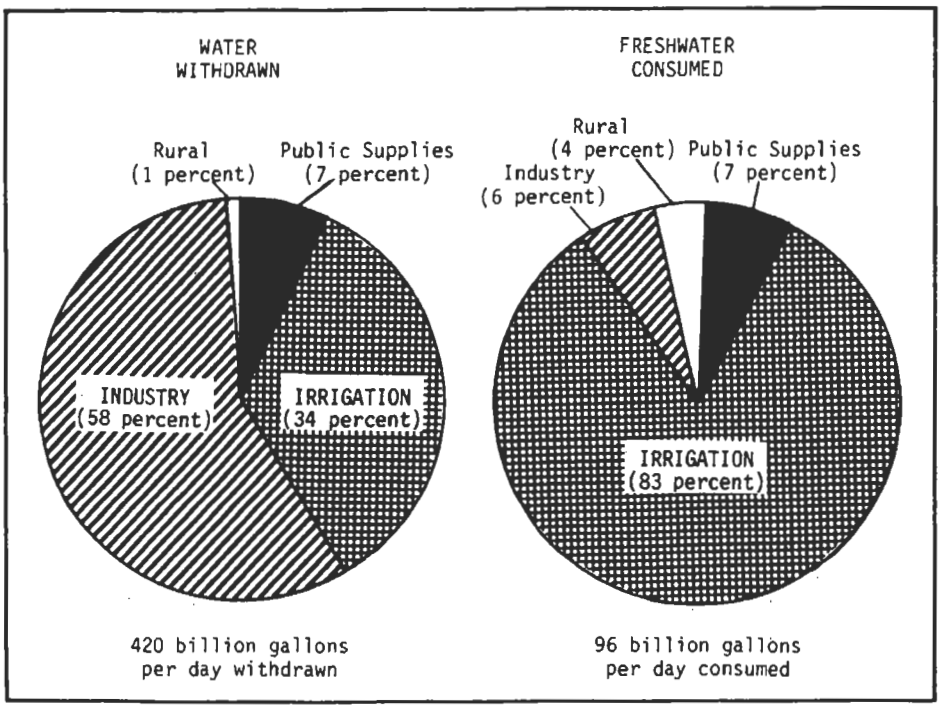

Figure 1. Off-channel water withdrawals in the United States and the proportion of freshwater consumed in 1975 (after Murray and Reeves 1977).

Because US society as a whole is such a large consumer of goods, it is easily understood why industrial water use per capita is high (Figure 1). Industry consumes about 10% of the total amount of water it withdraws (USWRC 1979). The amount of water used to produce different products is varied, but often is surprisingly large. For example, the production of an automobile requires 45,000–61,000 l of water and a single glass bottle requires 450–2500 l (Chanlett 1979).

Manufacturing industries withdraw about 193 bld of water (USWRC 1979). Almost 90% of this water is associated with industries located in the eastern portion of the United States. Fortunately, the water is recycled and reused within the plants an average of 2.2 times before it is discharged (USWRC 1979). Although pollution controls undoubtedly will encourage water recycling within industrial plants, water consumption is still expected to more than double by the year 2000 (USWRC 1979).

Water Demand for Food Production

Water is one of the major limiting factors for agricultural production. Although sufficient rain falls upon eastern US agricultural land, periodic drought continues to limit yields as was evident in the summer of 1980. Crops require and transpire massive amounts of water; for example, a corn crop that produces 5600 kg of grain per hectare will take up and transpire about 2.4 million liters of water per hectare during the growing season (Penman 1970). Irrigated crop production requires large quantities of water. For example, the production of 1 kg of the following food and fiber products under irrigation in California requires: 1400 l for corn, 1900 l for sugar (sugar beets), 4700 l for rice, and 17,000 l for cotton (Ritschard and Tsao 1978). Water needed to produce 1 kg of grain-fed meat ranges from about 4200 to 8300 l when the water input for irrigated grain is included.

In total, agriculture withdraws 600 bld annually (USDA 1980a). Irrigation accounts for 99% of this total, whereas livestock uses only 1%. In the nation as a whole, surface water supplies about 60% of the water used in irrigation and groundwater provides the remainder (Murray and Reeves 1977); however, major regional variations exist with some states pumping as much as 94% of their irrigation water from groundwater (Dvoskin and Heady 1976). About 16% of the irrigation water does not reach the

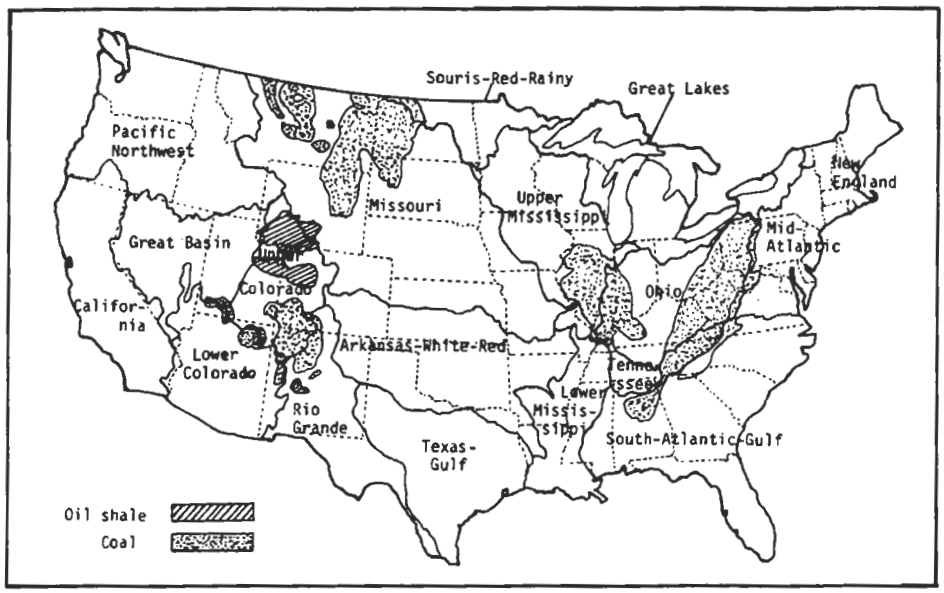

Figure 2. Map of the conterminous United States showing Water Resources Council regions and major coal and oil shale deposits (after Harte and El-Gasseir 1978, USWRC 1979).

intended crop; about one-third of this is lost by evaporation and the remainder by seepage and percolation during transport (Murray and Reeves 1977).

Although only about 12% of US cropland is irrigated, this land produces about 27% of the total dollar value of US crops (USDA 1980b). Most irrigated crop production is regionally concentrated. Seventeen western states contain nearly 90% of the irrigation hectarage in the United States (USDI 1979) and use about 93% of the irrigation water that is withdrawn in the nation (USWRC 1979). Considerable water must be supplied to crops grown in arid regions, and water demand is increased in these regions by high evapotranspiration rates. For instance, from 20–40% more water is used by crops grown under arid conditions than under normal moist conditions[1] (Arkley 1963).

In addition to water use, irrigated crop production is costly in terms of energy. The total energy used to supply each person annually with food is about 1500 l of oil equivalents (17% of per capita energy consumption) (Pimentel and Pimentel 1979). Nearly one-fifth of all energy consumed in agricultural production moves irrigation water (USDA 1974). In Nebraska, rain-fed corn production requires about 630 l/ha of oil equivalents, whereas irrigated corn requires about 1900 l/ha of oil equivalents or almost three times more energy (Pimentel and Burgess 1980).

The heavily subsidized federal and state-sponsored irrigation-water projects also consume large quantities of energy in addition to that used by farmers for irrigation (Anonymous 1982). Significant energy inputs for irrigation are hidden in subsidized dam and waterway construction; energy inputs for such construction may account for an additional 10% of the energy used for surface irrigation water supply (Roberts and Hagan 1975). In addition, significant quantities of energy are used to lift water over mountain ranges in some regions of the West. In the San Diego River basin, for example, the surface water supplied through the Colorado River Aqueduct requires about 185 l of oil per 1.2 million l for this lift (Roberts and Hagan 1977). This increases from 2- to 10-fold the total energy input for irrigation water in some areas.

The total cost of energy used to pump irrigation water in the United States rose from about $350 million in 1973 to more than $1 billion in 1977 and represents a nearly three-fold increase (Sloggett 1979, USBC 1979). In certain locations where groundwater depth is great, the energy costs of pumping water may be another limitation to irrigation.

In the West surface water for irrigation ranges from $10 to $15 per million liters (OTA 1980b), whereas groundwater costs about $30 per million liters (USDA 1981). Pumping groundwater from depths of 180 m increases the cost to about $60. If 3.6 million liters per hectare of groundwater were used during the growing season, then the cost of water alone might range as high as $450/ha. Therefore, as fuel prices escalate, irrigated agriculture will face economic difficulties. This pressure is already being felt. For example, in the Trans Pecos region of Texas, because of the increase in irrigation and other production costs, crops like alfalfa, barley, sorghum, and wheat returned less than the breakeven price (Patton and Lacewell 1977). Also, in Arizona, the high price of fuels for irrigation is forcing some growers to abandon the production of low-value crops like alfalfa (Larson and Fangmeier 1977).

The quantities of irrigation water applied to cropland can be reduced by employing various technologies. Surface application uses the largest quantity of water and drip irrigation the smallest amount (Batty and Keller 1980). Drip irrigation delivers the needed amount of water directly to each individual crop plant, but piping the water to each plant is energy costly in terms of capital equipment. Drip irrigation, however, saves energy where high lifts are involved because pumping energies are much larger than installation energies (Batty and Keller 1980). Clearly, several opportunities exist to improve irrigation technologies to conserve water and energy (Batty and Keller 1980, Chen et al. 1976, Stanhill 1979). Future shifts in irrigation technology and crops will probably reduce both water and energy use per irrigated crop hectare.

Table 1. Per capita water withdrawals and water consumed in the eastern and western Water Resources Council regions (Figure 2) of the United States in 1975 (Murray and Reeves 1977).

	Public water supplies			Totals	
	Population served* (millions)	Per capita withdrawals (l/day)	Total pop. (millions)	Per capita withdrawals (l/day)	Fresh water consumed (l/day)
Eastern region	120.9	610	151.7	5700	370
Western region	50.8	720	61.3	11,400	4900
50 states	172.7	640	214.2	7200	1670

*Remaining population is served by private water supplies primarily from wells.

[1]L. D. Swindale, ICRISAT, Hyderabad, India, personal communication, 1980.

Water Demand for Energy production

During 1975, electric power plants withdrew a calculated 520 bld, 33% of the nation's total water withdrawals; this amount is projected to increase to about 760 bld by the year 2020 (USWRC 1979). Although power plants pump a large proportion of the country's water, they are responsible for only 2% of the total national water consumption, since most of the water is returned (Loftness 1978).

As US oil and natural gas reserves continue to decline during the coming decades, a shift to coal-derivatives and oil shale will probably occur to supply the nation's energy needs (Livingston et al. 1981, NAS 1979). However, this shift to synthetic fuel use faces several obstacles. One major problem is the large quantity of water needed for the production of synthetic fuels from coal and oil shale. Although synfuel production technology is still in the developmental stage and exact water requirements are unknown, the estimates require large quantities of water. Water is used in mining coal and shale, in the conversion process to high-grade fuel (oil or gas), and in waste treatment and land reclamation (Table 2). For example, 25–75 l of water per metric ton of mined coal are used for dust control and washing the mined coal. In addition, coal liquefaction consumes 1200–7500 l per barrel (42 gal or 159 l) of synfuel produced (Table 2). The water consumed in synfuel production from coal includes water used for the following purposes: mining (1%), reclamation (15%), transport of slurry in pipelines (15%), conversion process (60%), and associated urban uses (9%) (Harte and El-Gasseir 1978, OTA 1978).

Using coal to produce synthane gas consumes 620–3900 l of water per million kilocalories of synthane produced (Table 2). Converting coal energy into gas or syncrude is about 68% efficient (NAS 1979). Thus, each metric ton of converted coal produces the equivalent of about 900 l (5.7 bbl) of oil and consumes 870–7400 l of water.

The production of oil from shale consumes less water than producing synfuels from coal (Table 2). The energy potential, however, of oil shale is estimated to be much less than synfuels from coal (NAS 1979). The water consumed per million kcal of oil from shale is calculated to be 470 l, or 680 l per barrel of oil (Table 2) and the largest water input (46%) is for land reclamation (Harte and El-Gasseir 1978). Such a high water requirement for reclamation is due to the difficulty in revegetating the large areas of barren rubble that result from processed shale. The volume of processed shale is about 1.2 times greater than the raw shale, creating a major environmental problem.

Earlier, it was mentioned that agriculture uses about 300 bld of the total 360 bld water consumed and that the highest use occurs in the western United States. Therefore, a major synfuel energy conversion program using coal or oil shale to produce oil and/or gas may seriously compete with agriculture for water in western regions of the country (Bishop and Narayanan 1979) (Figure 2). Energy producers will have significant difficulty in securing a share of the western water where agriculture is the "major user of diverted water in the West, accounting for 90% of water consumption" (Radosevich 1979, p. 3). Western water laws are based on the concept of "first in time, first in right" instead of riparian access, as in the East. Currently most western water rights are allocated to agriculture, primarily for irrigation.

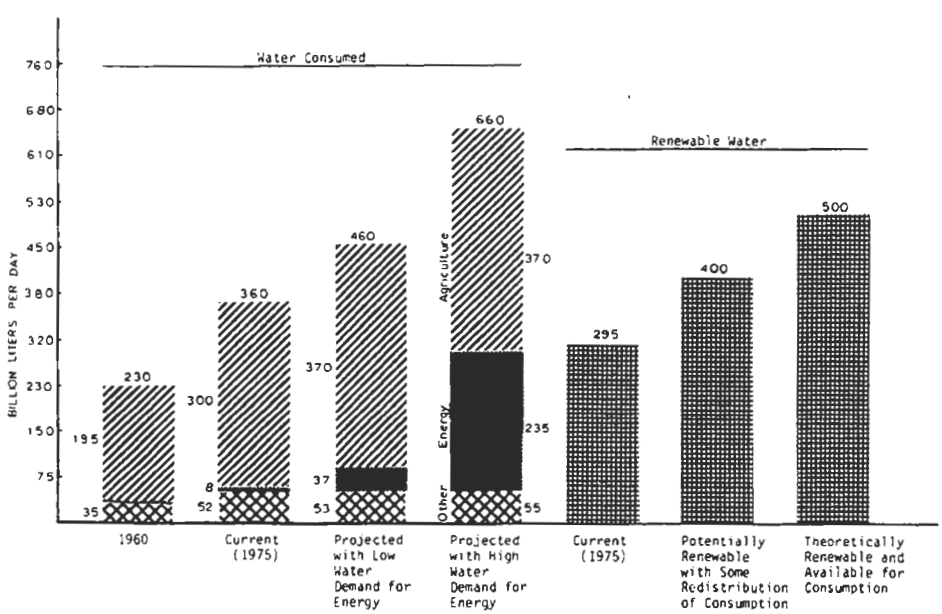

Figure 3. Water consumption levels for 1960 and 1975 by agricultural and energy producers and others (Murray and Reeves 1977, USWRC 1979). The projected water consumption levels after 2000 with proposed agricultural and energy programs are indicated. In addition, the various renewable water projections are shown; it should be noted that none of these estimates of renewable water resources would result in ground-water mining or depletion of minimal stream flows for wildlife and aquatic ecosystems (USWRC 1979).

WATER USE AND AQUATIC ECOSYSTEMS

Many aquatic ecosystems have been either destroyed or are now under stress from competing water needs for agriculture, energy, domestic, and industrial purposes. In areas of the West, for instance, when more than 40% of the mean annual flow of instream water has been depleted, a suitable habitat for most aquatic life cannot be maintained (Bayha 1976, Tennant 1976). Examples of such problem areas are the lower Colorado River, where nearly 80% of the instream flow has been depleted, and most ecosystems in southern California, Nevada, Arizona, New Mexico, Colorado, and western Texas, where about 70% of the instream flow has been depleted (USWRC 1979).

Aquatic ecosystems are also affected by heavy pumping of groundwater (CEQ 1980b), which is associated with salt water intrusion into the freshwater reserve and is occurring on Long Island and along the coast of California, Georgia, and Florida (Anderson and Berkebile 1976, Hammer and Elser 1980, West 1980). Another major problem associated with groundwater overdraft is "subsidence," that is, the settling of the land as water is pumped from the ground. In some regions, such as San Joaquin Valley, California, where irrigation pumping is heavy, the land already has settled 9 m (Bouwer 1977). This subsidence is irreversible once it occurs and may result in sufficient stress to cause faulting similar to the San Joaquin Valley and elsewhere (Kreitler 1977).

Special pollution problems result from irrigated agriculture when river and

Table 2. Water requirements for producing energy as coal and synthetic fuels.

Energy source	Consumption for water	Water needed ($l/10^6$ kcal)	Uses included	Source
Western coal mining	25–62 l/t*	4–9	Dust control, coal washing	USWRC 1974
Eastern surface mining	67–75 l/t	10–11	Dust control, coal washing	USWRC 1974
Oil shale (mine retort)	680 l/bbl	470	Mining and processing	EPA 1979
Coal gasification (synthane)	6×10^6–36×10^6 l/MCM†	620–3900	Mining, processing, and reclamation	Gehrs et al. 1981
Coal liquefaction	1200–7500 l/bbl	810–5100	Mining, processing, and reclamation	Gehrs et al. 1981

*metric tons
†million cubic meters

stream water is degraded by the addition of salts. For example, when the Colorado River flows through Grand Valley and water is withdrawn for irrigation and later returned to the river, an estimated 18 t/ha of salt are leached from the irrigated land and added to the river water (EPA 1976). At times during the summer, the Red River in Texas and Oklahoma is more saline than seawater, mainly due to irrigation use and normal evaporation (USWRC 1979).

Energy production from coal and shale may also increase water pollution. Water pollution from coal and shale oil comes from point and nonpoint sources, including cooling water discharges, spills, pipeline leaks, and leaching of the aboveground waste disposals (Gehrs et al. 1981, OTA 1980b). Spent shale in aboveground retorting will probably be leached by rainfall, and this alkaline residue contains significant amounts of sulfate, carbonate, and bicarbonate, as well as traces of other organic compounds (OTA 1980b).

From mining to electrical generation, the use of coal can also result in pollution problems. Acid runoff from mines and spoil banks may pollute nearby streams and watersheds. Coal mining may also reduce groundwater supplies and pollute groundwater through acid drainage (Elliot 1981). Revegetation of the strip-mined land is often difficult. In addition, coal combustion is a major source of air pollutants including particulates and oxides of sulfur and nitrogen. The large amount of sulfur dioxide and nitrous oxides in the atmosphere results in "acid rain" (NAS 1981). The ecological effects of acid rain include toxic conditions for some beneficial plants, including crops, insects, and fish.

Surface and groundwater pollution is clearly a hazard to the environment and public health. In addition, economic costs associated with cleaning water for use and reuse are significant. For instance, the costs of conventional treatment of potable water normally range from \$0.11 to \$0.22 per 1000 l (Clark 1979), but may be as high as \$0.33 per 1000 l[2]. If salts must be removed, the costs are about \$0.21 (EPA 1980) and the treatment of sewage for release into streams and lakes costs about \$0.08 per 1000 l (EPA 1980). Thus, environmental sensitivities to pollution are expected to be important constraints to future water development or agricultural and energy production.

PROJECTIONS AND CONCLUSIONS

Both water withdrawals and consumption have been growing steadily since 1950 and are projected to increase each year in the future (USC 1979, USDA 1980a). Some sources project a slight decline in fresh water withdrawals but an increase in overall consumption (USWRC 1979). From 1950 to 1975 per capita water withdrawals increased more than 45% (Murray and Reeves 1977). From 1960 to 1975 water consumption increased 57%, and the western region experienced the major increase in this growth (Figure 3).

Water consumption by industry and the general public will rise, but the major growth in water consumption is projected for agricultural and energy production. Agricultural use of water will grow because of a projected 30% increase in crop and livestock production by the year 2000 in the United States (ERAB 1981). Some factors contributing to agricultural growth include: increased food exports (estimated to be \$45 billion for 1981) to help offset the nation's oil imports, the rise in per capita use of food crops (USDA 1980c), and US population growth (USBC 1979). Projections are that fresh water consumption in agriculture will increase about 17% from 1975 to 2000 (USWRC 1979).

In addition to agricultural needs, liquid and gas synfuel production from coal and shale oil resources will require large inputs of water in the future with major development probably not taking place until after the year 2000. To put synfuel production from coal into perspective, assume that the nation moved to meet its total current oil and gas needs by coal conversion. Based on the available data associated with synfuel production from coal (Table 2) and producing 10×10^{15} kcal of oil and 5×10^{15} kcal of gas (DOE 1978), increased water consumption is calculated to range from 30 to 230 bld (Figure 3). The nation is currently consuming 360 bld, therefore water consumption would increase 8–64%, and a significant percentage of this could be expected to occur in the West where water shortages already exist. Another option is to transport the coal to locations where water is available for processing for synfuel production; however, this raises the energy costs.

If the projected growth in agricultural and synfuel programs is combined, the total water consumption increase could be 83–280 bld by the year 2000. With these scenarios, projected total water consumption by the nation might range from 460 to 660 bld (Figure 3). Note that both of these projections are greater than the quantity of water (400 bld) that is potentially renewable with some redistribution of consumption in the nation. The projected high water demand for energy production is greater than the quantity of water (500 bld) that is theoretically renewable and available for consumption. Increases of the magnitudes mentioned for energy and agricultural production can be expected to lead to serious conflicts in water use that will be especially intense in regions where water supplies are already short. Water consumption problems for the future can be better understood by examining current water deficits. Note that current water consumption is 360 bld, whereas current renewable water resources without groundwater mining and surface water depletion are estimated at only 295 bld (Figure 3).

All sectors of the US economy, however, are projected to continue to grow

[2]J. Rogers, Engineer, Village of Cayuga Heights, NY, personal communication, 1980.

and consume more water. The rates and patterns of growth of these sectors will probably be limited both by the dollar cost and availability of water. As a result, the current priority for water in agriculture in some regions could decline because of the economics of water use in crop and livestock production. For example, corn currently sells for about $0.11/kg and about 900 l of irrigation water may be required to produce a kilogram under arid conditions. Assuming that 1000 l of water cost only $0.025, then about $1 of water produces about $5 worth of corn (assuming no other input costs). Compare this to synfuel energy production where $1 of water produces a product worth about $350 (assume 4000 l of water is used to produce one barrel of oil worth $35 [Table 2]). The dramatic difference in economic value of product would enable energy producers to pay much higher prices for the essential water than farmers. Cost/benefit analyses will probably favor the more profitable industrial uses of water over agricultural uses (Richard 1978).

In some regions water tables will continue to decline and in turn pumping costs will rise. Pumping costs in some regions could well exceed any extra benefits of increased crop yields due to irrigation. Larson (1981) predicts that 1.4 million hectares of land irrigated from the Ogallala formation in the Great Plains will be converted back to dryland farming by the year 2000, and as a result food crop yields will probably be reduced 60% in these regions.

With the lower economic return on water for agricultural purposes, some displacement of water use from agriculture can be expected. This change is already occurring. For example, in Pinal County, Arizona, nearly 41,000 ha or "more than one-third of the area's irrigated acreage have gone out of production" (Wiegner 1979, p. 59). Trends like this will likely continue and will be primarily associated with low-value crops like alfalfa and other forages. Reduced forage production in the West where water supplies are already short could result in increased forage production in the eastern United States. Such a shift in forage production would probably be followed by a shift of some livestock production to the East.

The problem of water allocation is by no means a problem specific to the United States. As many as 80 other countries, which account for nearly 40% of the world population, now experience serious droughts (Kovda et al. 1978). Although the amount of water withdrawn per capita on a global basis is less than a third of US per capita use, the growth in world population can be expected to double water demand by the year 2000 (CEQ 1980a). Agricultural production is projected to consume an estimated 64% of total water withdrawn.

Not only will there be internal competition for water use, but this will extend to conflicts between nations that share common water supplies. In fact, about one-third of the major world river basins are shared by three or more countries (CEQ 1980a). Given current world food shortages and the additional pressures of rapidly growing human population, more water will be needed for both agriculture and energy production.

Next to sunlight and land itself, water is the most vital resource for agricultural production. In the next two decades, US agricultural production will probably expand about 30% to satisfy food needs for the nation and food exports, resulting in an estimated 17% increase in water consumption over current levels. If the nation then makes a major commitment to synfuel production after the year 2000, 8–64% more water would be needed. A significant portion of this growth in water demand is projected for the western states where most irrigated agriculture and one-half of US coal reserves are located. Water shortages and environmental limitations compound the problems. Because of water availability and distribution in the nation and possible conflicts arising among regions and programs, a national water policy based on both ecology and economics should be developed to insure equitable and effective use of water.

ACKNOWLEDGMENTS

We thank the following people for reading an earlier draft of this article and for their many helpful suggestions: William C. Ackermann, University of Illinois; J. Clair Batty, L. Douglas James, Utah State; A. M. Beeton, Great Lakes & Marine Waters Center, University of Michigan; William Dritschilo, UCLA; Richard Fluck, University of Florida; Walter Urban Garstka, Consultant; John Krummel, Oak Ridge National Laboratory; James E. Kundell, University of Georgia; Daniel Okun, University of North Carolina; Theodore M. Schad, Commission on Natural Resources, National Research Council; R. Eugene Turner, Louisiana State; Paul Weisz, James White, John F. Marshall, Mobil R&D Corp. And at Cornell University we thank W. Harry Everhart, Douglas Haith, C. A. S. Hall, Mitchell J. Lavine, Rachel Ledermann, D. Peter Loucks, J. Madras, R. T. Oglesby, Marcia Pimentel, and Ann Snook.

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Benefits and Risks of Genetic Engineering in Agriculture

Socioeconomic and environmental problems may be associated with transfer of traits

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Genetic engineering, or recombinant DNA (R-DNA) technology, offers many opportunities for improving agriculture and public health. This technology transcends classical plant and animal breeding by permitting the rapid transfer of genetic traits between entirely different organisms. Potential benefits include higher yields and enhanced nutritional value from crops and livestock, reductions in pesticide and fertilizer use, and improved control of soil and water pollutants. Nevertheless, some releases of genetically engineered organisms may have sobering ecological, social, and economic effects.

The scientific community's objective, therefore, should be to maximize the potential social and economic benefits from genetic engineering while minimizing the risks to public health, human welfare, and the environment. If a serious problem does result from the application of genetic engineering, it will hinder the future development of the technology.

Because we have had little experience in nature with genetically engineered organisms, diverse views exist on the ecology, genetics, population dynamics, and potential environmental and public health risks of these organisms. Some biologists claim that the risks of R-DNA technology differ little from those in classical plant and animal breeding and that modified organisms present no unique hazards to the environment (Brill 1988, NAS 1987a). Others state, however, that the ability to move genetic material from one organism into a totally different organism is sufficiently unique to warrant concern (Mellon 1988, Sharples 1983, Tiedje et al. 1989). For example, genes from cocoa trees have been incorporated into bacteria to produce cocoa extract (Wen and Kinsella 1987), and genes from bacteria have been moved into plants for insect control (Vaeck et al. 1987). Genetic characters from pigs and chickens have also been transferred into mice (NAS 1987b). Staunch opponents suggest that the hazards of R-DNA technology are so great that the technology should be abandoned (Rifkin 1983).

A serious problem arising from genetic engineering would hinder future development of the technology

In this article, we assess the potential benefits and risks of deliberately releasing genetically engineered organisms into the environment. We suggest some approaches and protocols that might be used to minimize the environmental risks associated with these releases.

Engineering crops for resistance

Crop resistance to pests. Engineering crop resistance to insect and plant pathogen pests offers opportunities to reduce the use of insecticides and fungicides in crop production. This approach can be expected to reduce problems from pesticides and improve the economics of pest control. Although crop resistance to pests generally offers environmental benefits, care must be exercised to avoid breeding toxic chemicals such as alkaloids into the crop or reducing the nutrient makeup (Pimentel et al. 1984).

Herbicide resistance in crops. Genetic engineering to create herbicide-resistant crops has the advantage of expanding the array of herbicide types for weed control (NAS 1987b). In some instances, herbicide resistance may make possible the use of a more effective herbicide, thus reducing the number of herbicide applications. However, engineering resistance to the relatively new, low-dosage (measured in grams/hectare) herbicides will not necessarily reduce environmental hazards (House et al. 1987, NAS 1987c). Also, there is a danger that increasing the number of herbicide-resistant crops will encourage wider herbicide use and will contribute to environmental problems.

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Ecological issues

Genetic transfer. Bacteria are capable of transferring novel DNA sequences to bacteria of other species and genera. Although these transfers are rare in nature (Trevors et al. 1987), the potential for transfer from engineered organisms designed for persistence in the environment is disconcerting (Strauss et al. 1986). For example, transconjugant bacteria that received plasmids considered to be well characterized have demonstrated "unexpected alterations" in chemical makeup, virulence, and antibiotic resistance (Stotzky and Babich 1984).

Moreover, genetic exchange may occur between closely related plants as well as in microorganisms. For example, important weed species have originated through the hybridization of two intrageneric species, such as the crosses of *Raphanus raphanistrum* × *R. sativus* (radish) and *Sorghum halepense* (Johnson grass) × *S. bicolor* (Sorghum corn) (Colwell et al. 1986).

Potential evolutionary changes in released organisms. An engineered organism may be genetically unstable (Lindow 1983a). Also, the additional genetic baggage may put the engineered organism at a competitive disadvantage to the unaltered organism in its native habitat (Alexander 1985). In most cases, genetic and/or ecological instability are likely to result in the exclusion of the engineered organism, but there is always a possibility that the added genes will be favored by natural selection, allowing the organism to become a pest. Such genetic and ecological changes have occurred in native as well as accidentally introduced species such as the European corn borer (Brindley et al. 1975).

Single-gene changes and pathogenicity. Most single-gene changes are not likely to affect adversely the pathogenicity and virulence of an organism in the environment (NAS 1987a). Some gene changes, however, may have detrimental consequences. Certain genetic alterations in animal and plant pathogens, for example, have led to enhanced virulence and increased resistance to pesticides and antibiotics (Alexander 1985). Genetic change has transformed a microbe from being commensal with its plant or animal host to being pathogenic (Leonard 1987).

For instance, two avirulent herpes simplex viruses were found to produce recombinants that were lethal to their host (Javier et al. 1986). In addition, a pathogen of grape with a limited host range was converted to a strain with a wide host range when a single gene was transferred to it (Kerr 1987). Also, some oat rust microorganisms, initially nonpest genotypes for a particular oat variety, became serious pest genotypes after a single gene change allowed the rust to overcome resistance in the oat (Browning 1974). This phenomenon has led plant pathologists to develop the "gene-for-gene" principle of parasite-host relationships, where a single mutation in a parasite overcomes single-gene resistance in the host (Person 1959).

Support for the principle that small genetic differences can be ecologically important was provided by Brenner's (1984) conclusion that *Escherichia coli,* the generally commensal species found in human digestive tracts, and all described species of *Shigella* could be considered one species on the basis of DNA similarity. *Shigella,* however, is invariably pathogenic, and *Shigella dysenteriae* was responsible for an epidemic of dysentery in Mexico and Central America in which 500,000 cases were reported with a fatality rate as high as 35% (Sharples 1983).

Furthermore, numerous instances have been documented in which insects, through a single gene change, have overcome resistance in plant hosts or have evolved resistance to insecticides (Gallun 1977, Roush and McKenzie 1987). At least 447 species of arthropods have developed resistance to pesticides (Roush and McKenzie 1987).

Dangers from modified native organisms. Lindow (1983b) reported that there is little or no danger from the ice-minus strain of *Pseudomonas syringae* (Ps), because Ps is a native organism that produces related phenotypes in nature. Because some native organisms have the ability to alter their interactions within an ecosystem, the genetic modification and release of native species into the environment may not always be safe.

For example, approximately 60% of the major insect pests of US crops were once harmless native organisms (Pimentel 1987). Many of these insects moved from benign feeding on natural vegetation to destructive feeding on introduced crops. The Colorado potato beetle moved from feeding on wild sandbur to feeding on the introduced potato (Casagrande 1987). This beetle has become a serious pest, and the damage has subsequently encouraged heavy use of insecticides.

Similarly, nearly half of the major weed species in US agriculture are native plants that have invaded crop habitats (Pimentel 1987). Approximately 30% of the plant pathogens on US crops are native microorganisms that are also parasitic on native vegetation (Pimentel 1987). Therefore, native organisms are not necessarily harmless to agriculture.

Dispersal and movement. The natural dispersal of a pest organism in the environment depends on many factors (Andow 1986). For example, the westward movement of Dutch elm disease fungus from the east coast of the United States has been relatively slow due to prevailing westerly winds. These winds have also impeded the movement of both the European and American bark beetles, carriers of the fungus (Sinclair and Campana 1978). Conversely, northerly winds facilitate migrations of the potato leafhopper and the true armyworm. These insects travel each spring from the southern United States to crops in the Northeast and Midwest, a distance of approximately 2500 km (McNeil 1985).

Microorganisms, including plant pathogens, are also dispersed by wind. A population of *Bacillus* has been carried by wind from points of the Black Sea to Sweden (1800 km; Andow 1986), and wheat stem rust is transported approximately 2500 km northward each year in the United States (Roelfs 1985).

Once a modified organism is released, therefore, its dispersal will be difficult to monitor effectively and to control. In fact, only two microorganisms have been controlled after introduction: the human disease organism smallpox and the plant

pathogen citrus canker (reintroduced in 1984).[1] The only macroorganism that has been exterminated from the United States after introduction has been the Mediterranean fruit fly (Hagen et al. 1981). This medfly has been exterminated five times from Florida, once from Texas, and once from California.

Probability of environmental risks. The probability of an environmental problem occurring after a single release of any engineered organism cannot now be accurately predicted. Some ecologists and genetic engineers suggest the risk is low (Alexander 1985, NAS 1987a, OTA 1988, Tiedje et al. 1989). Successful early releases may create the public perception that R-DNA organisms are risk-free. The probability of a problem occurring will increase, however, if public confidence leads to relaxation of the protocols. Commercialization, large-scale releases, and multiple releases in diverse habitats will also increase the probability of problems.

One might also expect the potential environmental problems caused by the release of genetically engineered organisms to vary widely in severity. Experience has shown that exotic organisms have a wide range of impacts on ecosystems. In some instances, exotic species may displace native species but assume a similar role in the community. Native species of Hawaiian talitrid sandhoppers, for example, were displaced by an exotic amphipod sandhopper species, *Talitroides topitotum* (Howarth 1985). In other cases, exotic species may place new stresses on the ecosystem; the devastating impact of the introduced European gypsy moth on trees and shrubs is an example (Cameron 1986).

In spite of the rigorous US government plant and animal quarantine program and the low survival rate of introduced foreign organisms, some pest organisms have become established in the United States (Pimentel 1987). Recent examples are the reintroductions of the Mediterranean fruit fly into California during the summers of 1987 and 1988.[2] The last medfly eradication effort in California required massive amounts of insecticides, costing the government and farmers a total of $174 million (Jackson and Lee 1985). Major pests also cost the United States $64 billion annually in crop and livestock destruction, despite the annual application of approximately 500,000 tons of pesticides (Pimentel 1986a).

The potential costs of damage resulting from a new pest introduced via genetic engineering, or other causes, can be estimated from data on some current US pests. For example, corn rootworms cost the United States approximately $2 billion annually (Pimentel et al. 1988). Similarly, the European gypsy moth causes an estimated $100 million in damage to ornamental trees and shrubs and commercial forests, and it costs the United States an additional $10 million for control each year (Pimentel et al. 1988).

Although the public appears willing to accept a risk of 1 chance in 1000 for the occurrence of an environmental problem (OTA 1987), one release could result in a disaster that could rapidly change the public's attitude toward genetic engineering (Panem 1985).

Intentional introduction of crop plants and animals. Some proponents of R-DNA technology suggest that the intentional introduction of foreign plants and animals into the United States is a good model for predicting potential problems from genetic engineering (NAS 1987a). If so, there is reason for concern because some serious problems have resulted from the intentional introduction of what were believed to have been beneficial organisms. Genetic similarities between many crops and weeds are evident from the fact that 11 of the 18 most serious weeds of the world are crops in other regions of the globe (Colwell et al. 1985). In the United States, for example, of a total of 5800 introduced crops, 128 species of agricultural and ornamental plants have become pest weeds (Table 1).

Furthermore, nine out of a total of twenty introduced domestic animal species in the United States have displaced or destroyed native species and, in general, have become serious environmental pests (Table 1). Similarly, a total of five introduced fish species have become pests. Of an estimated 2000 fish species that have been introduced and cultured (McCann 1984), 104 species have been detected in streams and lakes in the continental United States and 41 species have become established, 16 of which have expanding populations (Courtenay et al. 1984). Although many of these established species have not been declared to be pests, there is evidence that many are displacing native species. Ten other introduced mammal species, including the mongoose and the wild boar, as well as four bird species, have also become serious pests.

This tally need not suggest that all introductions should stop. Specialist predators and parasites of pest insects

Table 1. Introductions of agricultural and ornamental plants and agricultural, sport, and pet animals that became pests* in the United States (Pimentel et al. 1988).

Introductions	Number of species intentionally introduced	Number of species that have become pests
Agricultural and ornamental plants	5800†	128
Domestic mammals and birds	20‡	10
Sport mammals and birds	20§	9
Biological control vertebrates	5	5
Aquarium and sport fish	2000‖	5

*A pest can be narrowly defined as an organism with direct negative impact on human welfare or can be more broadly defined to include negative impacts on indigenous organisms and habitats. Many of the authors cited used the narrow definition.
†S. Kresovich, 1987, personal communication. USDA-ARS Germplasm Resource Unit, Geneva, NY.
‡Estimated number of introduced mammals and birds.
§Estimated; however, this value does not include 51 exotic species introduced into Texas for game purposes, none of which to date have been released or escaped into nature (Armstrong and Wardroup 1980).
‖Estimated (McCann 1984).

[1]E. Civerolo, 1988, personal communication. USDA-ARS, Beltsville, MD.
[2]M. Holmes, 1987, personal communication. APHIS, USDA, Beltsville, MD.

and weeds introduced for the purposes of biological control have generally had minimal environmental impacts and have reduced the need for environmentally disruptive pesticide applications. Whereas a few biocontrol agents became pests in the early 1800s and 1900s, greater ecological knowledge and established regulations have reduced the risk of these introductions (Pimentel et al. 1984).

This history suggests that the introduction of many types of foreign organisms in the environment could have a major negative impact on many of the 200,000 beneficial plant and animal species in the United States (Pimentel et al. 1980).

Predicting ecological effects. An important step in identifying the environmental problems associated with R-DNA technology is predicting on a case-by-case basis the potential ecological effects of releasing an engineered organism. Ecologists can predict with a high degree of accuracy the survival rate and interrelationships among some species populations in some environments. For example, insects introduced from the humid tropics have one chance in ten of surviving in most ecosystems of temperate Europe and North America (Williamson and Brown 1986). Ecologists can also predict with near certainty that a pest insect introduced from Europe will become a pest in the United States if the crop it feeds on in Europe is also widely grown here.

However, ecologists cannot currently predict with the same degree of accuracy the ecological impacts of all released organisms. For example, of the 212 introduced insects that have become major pests in the United States, 65% were not pests in their native ecosystems (Calkins 1983). In these cases, predictions based on the ecology of the insects in their natural habitat would have been inaccurate.

Many scientists suggest that each engineered organism be evaluated individually, before release, by focusing on the ecology of the unaltered organism and on the proposed release environment (NAS 1987a, Stotzky and Babich 1984). As more information becomes available on the environment and the biology of each species, greater accuracy in ecological predictions will become possible.

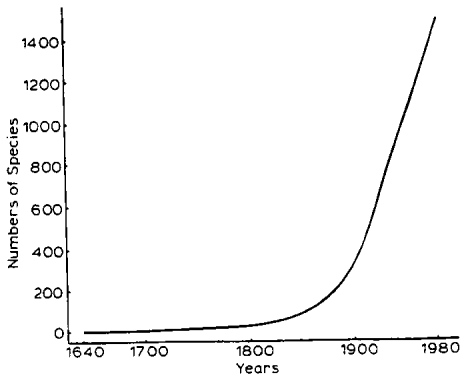

Figure 1. Number of species of insects and mites accidentally and deliberately introduced into the 48 contiguous US states from 1640 to 1980 (after Sailer 1983).

Ecological niches. An estimated 1500 exotic insects have become established in the United States with a few (17%) becoming pests or having an impact on native species (Sailer 1978; Figure 1). This observation indicates that few of the niches in natural ecosystems are filled (Colwell et al. 1985, Herbold and Moyle 1986), and there is ample opportunity, therefore, for many new species to become established in the United States. Thus, although the argument that engineered organisms will not become established due to competition with native species may apply in some cases, it is not always valid.

Economic value of biological diversity. There are many economically valuable species of plants, animals, and microorganisms. Uncultivated wild plants, for example, contribute raw materials for drugs, medicinals, and other items worth $40 billion per year in the world pharmaceutical market (Myers 1983). The crop germ plasm used by plant breeders to improve crop varieties has been estimated to result in an increase in the value of US crops of approximately $1 billion annually (Duvick 1984). Naturally occurring microbes with unexpected abilities to degrade man-made chemical pollutants have also recently been identified (Roberts 1987). Furthermore, introduced predatory and parasitic insects save California agriculture alone nearly $1 billion per year (van den Bosch et al. 1982).

Due to worldwide habitat destruction, however, biological diversity is rapidly decreasing, with extinctions occurring at a rate of approximately one species per day (Myers 1983). The advent of R-DNA technology offers some hope for the preservation of biological diversity, because the ability to extract genetic information from virtually any organism enhances the economic value of diversity (Brill 1979). However, the possible displacement of native species by released genetically engineered organisms may reduce biological diversity in ecosystems.

Preservation of genetic diversity in agriculture. Traditional plant breeding techniques have dramatically reduced genetic diversity in most crops. Unfortunately, this genetic uniformity has increased crop vulnerability to insect pests, diseases, and climatic fluctuations (NAS 1972). Genetic engineering, however, may potentially increase the diversity of crop plants through improved gene transfer and through germ-plasm screening and selection techniques (Hansen et al. 1986). Although most projects at this time involve single gene-transfers into crop plants, the feasibility of multiple-gene transfers is expected to improve. Yet some scientists suggest that genetic diversity in crops may not be achieved, and that long-term sustainability in agriculture will be reduced (Hansen et al. 1986). One of the reasons for this prediction is that every farmer desires to culture the highest-yielding new crop variety. Also, patents on some genetically engineered plants will inhibit the free exchange of germ plasm among breeders for development of new varieties, thus tending to restrict genetic diversity.[3]

The difficult ethical choices facing the research establishment are illustrated by the work on transferring the *Bacillus thuringiensis* (Bt) endotoxin into a number of crop plants to kill caterpillar pests (Vaeck et al. 1987). If the current plans for the use of Bt in crops are implemented, pest insects will be exposed to continuous high levels of the toxin in several crops, causing strong selection for resistant genotypes. This resistance could be

[3]M. E. Sorrells, 1988, personal communication. Cornell University, Ithaca, NY.

avoided, however, by producing crops that express the Bt toxin "only at times and in places where it is required" (Gould 1988). Linking the gene for the endotoxin with tissue-specific promoter sequences, for example, would be one way to protect valuable parts of a crop plant while leaving a susceptible refugium within the crop. These procedures, however, would be technically challenging and perhaps costly to implement.

Public policy issues

Risks of mismanagement. To realize the potential benefits of R-DNA technology, efforts must be made to prevent mismanagement. A National Academy of Sciences report (NAS 1987a) was criticized for including misleading summary statements (Colwell 1988) and has been said to be "particularly lacking in ecological perspective" (Mellon et al. 1987).

If the new technology is not carefully managed, substantial setbacks for the entire industry will occur. For example, after the Three Mile Island disaster, the entire nuclear industry lost credibility because of mismanagement at one Pennsylvania facility (Bignell and Fortune 1984). This accident resulted in the adoption of stricter regulations and the return to more expensive energy sources. Disasters (e.g., the Bhopal toxic chemical leak in India, the Challenger space-shuttle crash in the United States, and the Chernobyl nuclear disaster in the Soviet Union) in any heavily technological field can result in hostile public perceptions of advanced technologies (Slovic 1987). Unfortunately, several cases of mismanagement have already been associated with genetic engineering.

Mismanagement of a live vaccine developed for swine. A furor arose in 1986 when the US Department of Agriculture (USDA) approved the use of a genetically engineered, live-virus pseudorabies vaccine without following its own established approval procedures (Volkmer 1986). First, the developer did not receive permission to field test the vaccine as required by the National Institutes of Health Recombinant DNA Committee. Second, the USDA neglected to classify the vaccine as a recombinant organism. Finally, the USDA did not follow its own licensing procedures.

Although tests of this particular live vaccine have subsequently indicated that it is safe for swine and other mammals, some live vaccines for livestock may present environmental and public health problems. Microbes used in vaccine production, for example, can cause injury or death to humans and domestic animals (Meyer et al. 1971). This hazard has been demonstrated with both the live-vaccine polio virus and the rabies virus used in humans, which caused polio and rabies, respectively, in humans (Bijok et al. 1985, Melnick 1971). Thorough tests and meticulous care in following regulations, therefore, are necessary to prevent genetically engineered, live-microbe vaccines from causing the very diseases they are supposed to prevent.

Release and testing without approval. A rabies vaccine developed by the Wistar Institute of Philadelphia was tested in cattle in July 1986 by the Pan American Health Organization (PAHO) (Crawford 1987). However, PAHO failed to obtain Argentinian government approval for the trial, and the experiment was halted in November 1986. Some public health effects appear to have resulted from this release and are being investigated.[4]

During the spring of 1986, Advanced Genetic Sciences tested an ice-minus strain of *Pseudomonas syringae* in trees outside the greenhouse before receiving approval to do so from the Environmental Protection Agency (EPA) (Rogers et al. 1986). Ps is an important plant pathogen infecting 17 crops in California (Lindow 1982) and more than 100 species of wild plants (Lindow et al. 1978). Clearly, under these circumstances, tests in contained environments would be advisable to demonstrate that the modified strain was nonpathogenic to both crop and wild plants (Pimentel 1987).

More recently, a professor at Montana State University claimed to have released a genetically altered strain of Ps for control of Dutch elm disease. He had not received approval from EPA (Holdren 1987). Although the strain was later determined not to be an R-DNA strain, the professor's procedure introduced Dutch elm disease (in violation of the law) to Bozeman, Montana, a region previously free of this disease. Although such irresponsible behavior is certainly not common in the scientific community, this incident demonstrates the need for clear and enforceable regulations.

Socioeconomic value

Benefits. Genetic engineering could significantly improve yields and enhance the efficiency of crop and livestock production in the coming decades (NAS 1987b). These goals can be accomplished by increasing the proportion of a crop that can be harvested and by enhancing a crop's tolerance to various stresses. For example, modified Ps may reduce the susceptibility of certain crops to frost and, therefore, allow early planting. Similarly, nitrogen availability, a limiting factor in crop production, could be enhanced through R-DNA technology. Crops like corn and wheat might eventually be modified to fix their own nitrogen directly from the atmosphere, potentially saving the nation approximately $4 billion annually in fertilizer costs (Pimentel 1987).

In addition, organisms engineered to control pest insects, weeds, and plant pathogens could aid in reducing the more than $64 billion, or approximately 37%, loss of United States crop production due to pests (Pimentel 1986b). These modified organisms may potentially reduce a portion of the $4 billion spent on pesticide applications that not only destroy beneficial natural enemies, but cause other environmental, public health, and social problems (Pimentel 1987).

Costs. In contrast to the benefits of genetic engineering, significant social costs may be incurred (Krimsky 1987). For example, higher crop yields will benefit consumers by providing lower food prices, yet farmers' profit margins will generally decline. On average, for most crop and livestock products, a 1% increase in yield results in a 4.5% decrease in market

[4]D. Goldstein, 1988, personal communication. Universidad de Buenos Aires, Argentina.

price received by farmers.[5] This relationship can be illustrated with the case of bovine growth hormone (BGH).

Genetically engineered BGH has the potential to increase milk production in dairy cattle by as much as 40% (field estimates are closer to 10–25%; Rauch 1987). This hormone is currently under consideration for approval by the USDA and the US Food and Drug Administration (FDA). If put into use, BGH may dramatically increase milk production in the United States at a time when government purchases of surplus milk are up to 12.3 billion pounds per year (Rauch 1987). The expected 10% to 15% decrease in milk prices would probably further reduce the number of US dairy farmers (Kalter and Milligan 1988). All evidence suggests, therefore, that the use of BGH would accelerate the trend toward fewer and larger farms (Buttel 1988) and contribute further to the loss of cultural diversity in an increasingly urban society (Coen et al. 1987).

The potential effects of genetic engineering on rural land use have yet to be assessed. This rapidly advancing technology is, however, clearly capable of causing major ecological, economic, and social changes. Small farmers in developing countries, for example, may experience severe negative effects.

The use of microbes to synthetically produce cocoa, coffee, and tea extracts from relatively simple carbohydrates may eventually lead to the elimination of these industries in developing countries (Buttel and Barker 1985). For countries such as Ghana, where more than 20% of the work force is employed in cocoa production, this new technology could increase unemployment and reduce income from trade, in addition to stimulating unprecedented and environmentally destructive land use change, intensifying soil erosion and rapid water runoff (Christian 1985).

Because of the increasing significance of international trade, financial crises in developing countries can affect the economies of the developed world. This effect was clearly demonstrated in 1973–1974 when increases in oil prices caused US grain exports to decline 50% (USBC 1987).

Ethical issues. The financial rewards for successful research in genetic engineering are enormous. However, these incentives are unlikely to encourage innovation aimed at providing the greatest humanitarian good (Buttel et al. 1985). In addition, the highly competitive and secretive climate currently surrounding most genetic engineering research in the United States may slow the research process. For these reasons, a stronger public role in defining and supporting key research objectives, and in formulating standards to regulate the industry, is needed to temper the substantial influence of private enterprise on the development of R-DNA technology. Germ plasm, including genes such as those coding for Bt endotoxins, is a natural resource (NAS 1978), and as such merits government protection (Gould 1988).

Regulation of R-DNA technology

Government regulatory agencies. In June 1986, the federal government established guidelines for regulating the genetic engineering industry, in addition to creating an interagency system known as the Coordinated Framework for Regulation of Biotechnology (Federal Register 1986). Responsibility for controlling the safety of new products is now divided among five federal agencies (Committee on Science and Technology 1986):

- The USDA is responsible for engineered organisms used with crop plants and animals.
- The FDA is responsible for genetically engineered organisms or their products in processed foods and drugs.
- The National Institutes of Health (NIH) is responsible for engineered organisms that could affect public health.
- The Occupational Safety and Health Administration (OSHA) is responsible for engineered organisms that may affect the workplace.
- The EPA is responsible for engineered organisms released into the environment for pest and pollution control and related activities.

Divergent views exist regarding the merits of this pattern of industry regulation. Some observers believe these divisions of authority are cumbersome (Fox 1988), and others propose the abolishment of all the regulations (Szybalski 1985). Yet some argue that regulation actually assists the development of biotechnology (Gibbs and Greenhalgh 1983).

Agencies such as USDA, which both promote and regulate this new technology, may be faced with a conflict of interest. The same combination of promotion and regulation did not work when USDA was responsible for regulating the use of pesticides, and eventually control was transferred to EPA (GAO 1986). NIH has provided sound guidelines for laboratory research dealing with genetic engineering, but it is not a regulatory agency and, therefore, its only means of control is to seek EPA's help in regulation. EPA, however, is attempting to regulate genetically engineered organisms under the Toxic Substances Control Act and the Federal Insecticide, Fungicide and Rodenticide Act, but both acts have serious deficiencies (Schiffbauer 1985).

It is extremely important that fairness and effectiveness be achieved in regulatory policy (Committee on Science and Technology 1986). The needs are:

- The benefits and costs of R-DNA technology to society should be rigorously assessed. An attempt should be made to balance potential short-term benefits, both private and public, with possible long-term costs. Given this technology's potential social, economic, and environmental impacts, a prudent approach in minimizing these risks is needed. Public policies must cautiously yet fairly regulate R-DNA technology research and development.
- When establishing testing protocols, it is important to remember that the biotic and chemical interactions in nature are much more complex than those produced in laboratory tests. Currently, the safest approach is to include tests at progressively more complex levels of ecosystem structure. We propose

[5]D. B. Sisler, 1988, personal communication. Cornell University, Ithaca, NY.

the following steps: conduct laboratory microcosm tests to improve the understanding of the engineered organism; carry out small plastic-enclosed ecosystem field tests; and conduct small-scale field tests.

- Developing the means for tracking genetically engineered microorganisms is essential if the fate of these organisms in nature is to be determined. The monitoring methods must be specific, convenient, reliable, and sensitive (Omenn 1986). Currently, the six major monitoring methods are: selective media, resistance to particular antibiotics, immunofluorescence techniques, DNA genetic probes, R-DNA fingerprinting (Omenn 1986), and serological tests. To date, none of these is completely reliable or sufficiently sensitive. However, if more than one were employed, some of the deficiencies might be reduced.
- Various procedures have been suggested to control problems that might develop if engineered organisms are released into the environment. Some engineered organisms, like plants in pots, can be collected and destroyed relatively easily; however, others are more difficult to contain. Some microbes and macroorganisms, for example, disperse rapidly (Andow 1986). Others, like pox viruses, are extremely difficult to control by any technique.[6] Unfortunately, the record worldwide is poor for exterminating introduced pests after introduction (Joenje 1987).

Conclusions

Never before have scientists had the ability to transfer genetic traits between totally different microbes, plants, and animals. This ability presents unique risks to the environment, not unlike the problems created by introduction of exotic species. There are also socioeconomic problems associated with R-DNA technology. With only six field releases to date, there is little information or experience concerning potential environmental risks. As is common when data are insufficient, wide-ranging speculation and many misconceptions exist.

Based on our analyses, we conclude:

- Developing crops resistant to pests has minimal risks and should help reduce pesticide use.
- Although engineering herbicide resistance in crops may increase herbicide efficiency in some instances, this approach may encourage the use of a wider array of herbicides on a variety of crops—thus intensifying ecological problems associated with these pesticides.
- Although genetically engineered microbes released to attack specific chemical pollutants have the potential to improve the environment, these microbes must be compound-specific and produce harmless byproducts; if not, they may pose environmental hazards.
- When unique genetic characters are released into the environment via microbes and other organisms, a few of these novel DNA sequences may be transferred to other microbes and other organisms.
- Some single-gene changes may convert a benign organism into a serious plant or animal pathogen.
- Because most pest species are of native origin, the use of native organisms to genetically engineer new organisms is not without risk.
- Few ecological niches in ecosystems are completely filled; therefore, natural communities are unlikely to resist invasion by foreign organisms, including genetically engineered organisms.
- Based on past experience, the chance of an intentional foreign plant or animal introduction becoming a pest in the United States varies with the scale and frequency of the introductions from 1 in 10 to 1 in 100.
- R-DNA technology has the potential to increase genetic diversity in agriculture and forestry; however, due to numerous factors in development and use of crop and forest varieties, genetic diversity in these systems is expected to decline.
- Accuracy in predicting the ecological effects of releasing a genetically engineered organism depends on the specific organism, the type of genetic information introduced, the particular environment into which it is released, and the availability of detailed ecological information.
- Insufficient data exist to forecast environmental problems and pest outbreaks resulting from the release of genetically engineered organisms. Based on the data presented, we expect some environmental problems to occur because none of the test protocols will allow us to predict with 100% accuracy the impact of altered organisms on the environment. In the past, we have been unable to distinguish with total accuracy beneficial organisms from potential pests; a large number of releases into the environment increases the probability of a hazardous introduction, and the dangers caused by human error can never be eliminated.
- Several incidents have already demonstrated mismanagement of genetic engineering technology. A clear need exists, therefore, for more effective regulation and management of this technology.
- To achieve efficient, effective regulation, genetic engineering should be regulated only by EPA and OSHA. However, EPA should not attempt to carry out regulation under TSCA and FIFRA only; it needs new legislation.
- The socioeconomic costs and benefits of genetic engineering technology must be assessed in the process of formulating and implementing public policy.

Genetic engineering offers many opportunities for improving agriculture and public health. However, we believe the risks of this new technology must be confronted and effectively managed. Failure to exercise caution could lead to serious environmental, economic, and social problems in the United States and in all other nations. Although momentum is building for immediate results, potential problems could delay or jeopardize the realization of the potential benefits from genetic engineering.

Acknowledgments

We thank the following people for reading an earlier draft of this paper and making helpful suggestions: David Andow, University of Minnesota; Nyle Brady, US Agency for International Development (USAID); Winston J. Brill, Agracetus, WI; Liebe

[6]D. Goldstein, 1988, personal communication. Universidad de Buenos Aires, Argentina.

Cavalieri, Memorial Sloan-Kettering Cancer Center, New York City; Joel Cohen, USAID; Robert K. Colwell, University of California at Berkeley; Jack R. Coulson, ARS-USDA, Beltsville, MD; Charles Edquist, Linköping University, Sweden; Daniel J. Goldstein, Universidad de Buenos Aires, Argentina; Fred Gould, North Carolina State University; Arthur Kelman, University of Wisconsin; Stephen Kresovich, USDA-ARS Germplasm Resource Unit, Geneva; Sheldon Krimsky, Tufts University, MA; Joseph J. Molnar, Auburn University, AL; Norman Myers, Oxford, England; Zev Naveh, Israel Institute of Technology, Haifa; Gil Omenn, University of Washington, Seattle; A. Sasson, UNESCO, Paris; Guenther Stotzky, New York University; Trevor Suslow, Advanced Genetic Sciences, Oakland, CA; C. D. Upper, University of Wisconsin; M. Coburn Williams, Utah State University; and Yaffa Grossman, Ecological Society of America, Rockville, MD. At Cornell University, we thank: Fred Buttel, June Fessenden MacDonald, Robert Kalter, Richard T. Roush, Mark E. Sorrells, and Bruce Wilkins.

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Environmental and Economic Effects of Reducing Pesticide Use

A substantial reduction in pesticides might increase food costs only slightly

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Several studies suggest that it is technologically feasible to reduce pesticide use in the United States 35 to 50% without reducing crop yields (NAS 1989, OTA 1979). Denmark developed an action plan in 1985 to reduce pesticide use by 50% before 1997.[1] Sweden also approved a program in 1988 to reduce pesticide use by 50% within 5 years (NBA 1988). The Netherlands is developing a program to reduce pesticide use 50% in 10 years.[2] These proposals, along with the conclusion by Huffaker (1980) that the United States overuses pesticides, prompted us to investigate the feasibility of reducing the US annual use of synthetic organic pesticides by approximately one-half.

Farmers use an estimated 320 million kg (700 million lb) of pesticides annually at an approximate cost of $4.1 billion (Table 1). Indeed, investment in pesticide controls has been shown to provide significant economic benefits. Dollar returns for the direct benefits to farmers have been estimated to range from $3 to $5 for every $1 invested in the use of pesticides (Headley 1968, Pimentel et al. 1978). However, these cost figures do not reflect the indirect costs of pesticide chemical use such as human pesticide poisonings, reduction of fish and wildlife populations, livestock losses, destruction of susceptible crops and natural vegetation, honeybee losses, destruction of natural enemies, evolved pesticide resistance, and creation of secondary pest problems (Pimentel et al. 1980). Moreover, the economic benefits have been calculated for current agricultural practices, some of which actually increase pest problems. The direct and indirect benefits and costs of using pesticides in agriculture are complex.

There are many opportunities to reduce pesticide use

The objective of this article is to estimate the potential agricultural and environmental benefits and costs of reducing pesticide use by approximately 50% in the United States. To estimate the costs and benefits, we examine current pesticide use patterns in approximately 40 major US crops; evaluate current crop losses to pests; estimate the agricultural benefits and costs of reducing pesticide use by substituting currently available biological, cultural, and environmental pest-control technologies for some current pesticide-control practices; and assess the public health and environmental costs associated with reduced pesticide use.

Extent of pesticide use

Of the estimated 434 million kg of pesticides used annually in the United States, 69% are herbicides, 19% insecticides, and 12% fungicides (Table 1). The 320 million kg of pesticides used in agriculture are applied at an average rate of approximately 3 kg/ha to approximately 114 million ha—62% of the 185 million ha that are planted (Pimentel and Levitan 1986). Thus a significant portion (38%) of crops receives no pesticides.

The application of pesticides for pest control is not evenly distributed among crops. Overall, 93% of the hectarage of row crops, such as corn, soybeans, and cotton, is treated with some type of pesticide (Pimentel and Levitan 1986). In contrast, less than 10% of forage crop hectarage is treated.

Herbicides are currently being used on approximately 90 million ha in the United States—greater than half of the nation's cropland. Field corn alone accounts for 53% of agricultural herbicide use, and almost three-quarters of the herbicide is applied to corn and soybeans.

The unequal distribution is similar in insecticide use. Of the approximately 62 million kg of insecticides applied to 5% of the total agricultural land (Table 1), approximately 25% is used on cotton and corn. Some crops are treated as many as 20 times per

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[1]B. B. Mogensen, 1989, personal communication. National Environmental Research Institute, Copenhagen, Denmark.

[2]A. Pronk, 1990, personal communication. Wageningen Agricultural University, Wageningen, The Netherlands.

season (e.g., apples and cotton), whereas other crop hectares may be treated only once (e.g., corn and wheat).

Insecticide use also varies considerably among geographic regions. Warm regions of the United States often suffer intense pest problems. For example, although only 13% of the alfalfa hectarage in the United States is treated with insecticides, 89% of the alfalfa area in the Southern Plains states is treated to control insect pests (Eichers et al. 1978). In the Mountain region, where large quantities of potatoes are grown, 65% of the potato cropland receives insecticide treatment, but in the Southeast, where only early potatoes are grown, 100% of the potato cropland receives treatment (USDA 1975). Cotton insect pests such as the boll weevil are also more of a problem in the Southeast than in other regions (USDA 1983). In the Southeast and Delta states, 84% of the cotton cropland receives treatment, whereas in the Southern Plains region less than half of the crop (40%) is treated.

Fungicides are primarily used on fruit and vegetable crops (Pimentel and Levitan 1986). Approximately 95% of grapes and 97% of potato hectarage are treated with fungicides, whereas neither corn nor wheat hectarage is treated.

Crop losses and changes in agricultural technologies

Since 1945, the use of synthetic pesticides in the United States has grown 33-fold (Figure 1). The increase is largely due to changes in agricultural practices and cosmetic standards (Pimentel et al. 1977). At the same time, some new pesticides have at least tenfold greater effectiveness than older pesticides (Pimentel et al. 1991). For example, in 1945 DDT was applied at a rate of approximately 2 kg/ha. Today, similarly effective insect control is achieved with pyrethroids and aldicarb applied at 0.1 kg/ha and 0.05 kg/ha, respectively.

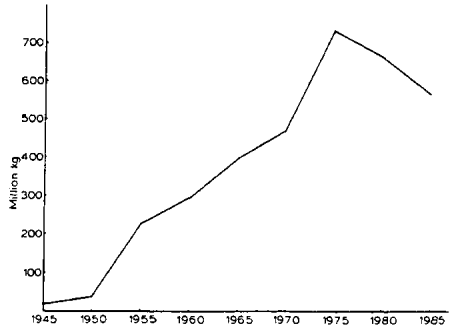

Figure 1. The amounts of synthetic pesticides (insecticides, herbicides, and fungicides) produced in the United States (Pimentel et al. 1991). Approximately 90% is sold in the United States. The decline in total amount produced is in large part due to the ten- to 100-fold increase in toxicity and effectiveness of new pesticides.

Currently, an estimated 37% of all crop production is lost annually to pests (13% to insects, 12% to plant pathogens, and 12% to weeds) in spite of the use of pesticides and nonchemical controls (Pimentel 1986). Although pesticide use has increased during the past four decades, crop losses have not shown a concurrent decline. According to survey data collected from 1942 to the present, losses from weeds have fluctuated with an overall slight decline, due to improved chemical, mechanical, and cultural weed-control practices, from 14% to 12% (Table 2). During that same period, US losses from plant pathogens, including nematodes, increased slightly, from 10.5% to approximately 12.0%. This increase results in part from reduced sanitation (because fungicides can substitute for sanitation), higher cosmetic standards, and abandonment of crop-rotation practices.

The share of crop yields lost to insects has nearly doubled during the last 40 years (Table 2), despite more than a tenfold increase in both the amount and toxicity of synthetic insecticide used (Pimentel et al. 1991). The increase in crop losses due to insects per hectare has been offset by increased crop yield obtained with higher-yielding varieties and greater use of fertilizers and other inputs, such as fertilizers, irrigation, and high-yielding crop varieties (USDA 1989).

This increase in crop losses despite intensified insecticide use is due to several major changes that have taken place in agricultural practices (Pimentel et al. 1991). These changes include:

- the planting of some crop varieties that are more susceptible to insect pests;
- the destruction of natural enemies of certain pests, which creates the need for additional pesticide treatments;
- the increase in pests resistant to pesticides;
- the reduction in crop-rotation practices;
- the increase in monocultures and the resultant reduced crop diversity;
- the lowering of Food and Drug Administration tolerance for insects and insect parts in foods and the enforcement of more stringent cosmetic standards by fruit and vegetable processors and retailers;
- the increased use of aircraft application technology;
- the reduction in sanitation, including less attention to the destruction of infected fruit and crop residues;
- the reduction in tillage, with more crop residues left on the land surface;

Table 1. US hectarage treated with pesticides (modified from Pimentel and Levitan 1986). Numbers are in millions.

		All pesticides		Herbicides		Insecticides		Fungicides	
Land-use category	Total hectares	Treated hectares	Quantity	Treated hectares	Quantity	Treated hectares	Quantity	Treated hectares	Quantity
Agricultural	472	114	320	86	220	22	62	4	38
Government and industrial	150	28	55	30	44	NA	11	NA	NA
Forest	290	2	4	2	3	<1	1	NA	NA
Household	4	4	55	3	26	3	25	1	4
Total	916	148	434	121	293	26	99	5	42

Total for hectarage treated with herbicides, insecticides, and fungicides exceeds total treated hectares because the same land area can be treated several times with several classes of chemicals. NA, not available.

- the planting of crops in climatic regions in which they are more susceptible to insect attack;
- the use of herbicides that have been found to alter the physiology of crop plants, making them more vulnerable to insect attack.

Estimated benefits/costs from reduced pesticide use

A reduction in US pesticide use would require substituting nonchemical alternatives for chemical pest control and improving the efficiency of pesticide application technology. Such changes in some cases increase and in other cases decrease pest control costs.

Crop losses to pests. Losses from pests for 40 major crops grown with pesticides have been estimated by examining data on current crop losses, by reviewing loss data based on experimental field tests, and by consulting pest control specialists. Combining these data, however, has often been difficult. For example, data based on published experimental field tests usually emphasize the benefits of pesticide use; thus loss data associated with pesticide treatments usually emphasize benefits over costs (Pimentel et al. 1978).

In addition, field tests often exaggerate total crop losses because assessments of insect, disease, and weed losses are carried out separately and then combined. For example, on untreated apples, insects were reported to cause a 50–100% crop loss, disease 50%–60%, and weeds 6% (Ahrens and Cramer 1985, Pimentel et al. 1991). This approach yields an estimated total loss of approximately 140% from all pests combined! A more accurate estimate of losses in the absence of pesticides ranges from 80% to 90% based on current cosmetic standards (Ahrens and Cramer 1985). Exactly how much overlap exists among insect, disease, and weed-loss figures for apples and for other crops is not known.

Our analysis has other important limitations. The figures for current crop losses to pests, despite pesticide use, are based primarily on US Department of Agriculture data and other estimates obtained from specialists. We emphasize that these are estimates. For certain crops, little or no experimental data are available concerning yields with pesticide use and various alternatives. In addition, in some cases recent data were not available. With these crops, our estimates were generally extrapolated from data on closely related crops.

Although we recognize the limitations of the data used in this analysis, we believe there is a need to assemble available information to provide a first approximation of the potential for reducing pesticide use by one-half. We hope that better data will be available in the future, so that a more complete analysis of pesticide costs and benefits can be made.

Reduction of the risks associated with pesticides is in itself a complicated issue, particularly because environmental and health trade-offs are often associated with changes in technology. Because of the complexity of these trade-offs, they could not be included in the analysis. One example, however, involves the conflict between reducing pesticide use and promoting soil conservation through the use of no-till culture as well as reduced tillage. Although no-till and reduced-till culture significantly reduce soil erosion (Van Doren et al. 1977), they also significantly increase the need for herbicides, insecticides, and fungicides (Taylor et al. 1984).

However, although reducing pesticide use may require reducing the use of some no-till systems, highly cost-effective soil conservation alternatives to no-till do in fact exist. These include ridge-till, crop rotations, strip cropping, contour planting, terracing, windbreaks, mulches, cover crops, and green mulches (Moldenhauer and Hudson 1988). Ridge-planting, for example planting the crop on permanent ridges 20 cm high along the contour, is rapidly growing in popularity as an effective replacement for no-till practices for most row crops. It is a form of reduced-till that has many advantages over no-till (Forcella and Lindstrom 1988). For example, ridge-till can be employed without the use of herbicides, and it controls soil erosion more effectively than no-till (Russnogle and Smith 1988).[3]

Techniques to reduce pesticide use. The increase in crop losses associated with recent changes in agricultural practices suggests that some alternative strategies exist that might reduce pesticide use. Two important practices that apply to all agricultural crops include widespread use of monitoring and improved application equipment. Currently, a significant number of pesticide treatments are applied unnecessarily and at improper times due to a lack of treat-when-necessary programs. Furthermore, pesticide is unnecessarily lost during application (e.g., only 25–50% of the pesticide applied by aircraft actually reaches the target area; Akesson and Yates 1984, Mazariegos 1985). By increasing monitoring and improving application equipment, more efficient pest control can be achieved.

Insecticides

Corn and cotton account for approximately 25% of the total insecticide use in agriculture. Thus reducing insecticide use in these two crops by substituting nonchemical alternatives would contribute significantly to a reduction in insecticide use.

Corn. During the early 1940s, little or no insecticide was applied to corn, and losses to insects were only 3.5%

Table 2. Comparison of annual pest losses* in the United States. (After Pimentel et al. 1990.)

Period	Percentage of crops lost to pests				Crop value lost (billion dollars)
	Insects	Diseases	Weeds	Total	
1989	13.0	12.0	12.0	37	150
1974	13.0	12.0	8.0	33.0	77
1951–1960	12.9	12.2	8.5	33.6	30
1942–1951	7.1	10.5	13.8	31.4	27
1910–1935	10.5	NA	NA	NA	6
1904	9.8	NA	NA	NA	4

*Not adjusted for inflation.
NA = Not available.

[3]R. Thompson, 1985, personal communication. Farmer, Boone, IA.

(USDA 1954). Since then, insecticide use on corn has grown more than 1000-fold, whereas losses due to insects have increased to 12% (Ridgway 1980). This increase in insecticide use and the 3.4-fold increase in corn losses to insects are primarily due to the abandonment of crop rotation (Pimentel et al. 1991). Today approximately 40% of US corn is grown continuously, with 11 million kg of insecticide applied annually (Pimentel et al. 1991). By reinstituting crop rotations, large reductions in pesticide use could be achieved. Rotating corn with soybeans or a similar high-value crop generally increases yields and net profits (Helmers et al. 1986), although rotating corn with wheat or other low-value crops reduces net profits per hectare. From a more comprehensive perspective, however, the rotation of corn with other crops has several advantages, including reducing weed and plant-pathogen losses and decreasing soil erosion and rapid water-runoff problems (Helmers et al. 1986).

By combining crop rotations with the planting of corn resistant to the corn borer and chinch bug, it would be possible to avoid 80% of insecticide use on corn while concurrently reducing insect losses (Schalk and Ratcliffe 1977). Such a move is estimated to increase the cost of corn production by $10 per hectare above the current costs of corn grown continuously (Pimentel et al. 1991). A new approach, in which an attractant combined with insecticides fights rootworm, has been reported to reduce insecticide requirements 99% (Paul 1989).[4]

Cotton. The potential for reducing pesticide use in US agriculture is well illustrated by changes in insecticide use in Texas cotton production. Since 1966, insecticide use in Texas cotton has been reduced by almost 90% (OTA 1979). The technologies adopted to reduce insecticide use were: monitoring pest and natural enemy populations to determine when to treat, biological control, host-plant resistance, stalk destruction (sanitation), uniform planting date, water management, fertilizer management, rotations, clean seed, and changed tillage practices (King et al. 1986, OTA 1979).

Currently, a total of 29 million kg of insecticide is applied to cotton, and it is estimated that this amount could be reduced by approximately 40% through the use of readily available technologies (Pimentel et al. 1991). By effectively using a monitoring program, one might reduce insecticide use by approximately 20%. Through the use of pest-resistant cotton varieties and the alteration of planting dates in most growing regions, insecticide use could be reduced by another 3% (Frans 1985, Frisbie 1985). An additional 10% reduction in insecticide use could be achieved through the replacement of the price-support program with a free-land market (without commodity and price-support programs) and no price-support program for cotton production (NAS 1989). This change would allow cotton to be grown in regions with fewer insect pests, thus reducing the need for insecticide use. However, changing the current price-support program to allow society to take advantage of these benefits is politically unattractive.

The amount of insecticide reaching target areas could be increased by changing the type of application equipment employed, especially reducing the use of aircraft ultra-low-volume application equipment, which wastes 75% of pesticide applied. The amount of insecticide waste could be reduced by 25% if ground-application instead of air-application equipment were used (Mazariegos 1985, Pimentel and Levitan 1986). In addition, covering the spray boom with a plastic shroud can further decrease drift 85% (Ford 1986), thereby allowing for an additional reduction in pesticide use.

Insecticide use on cotton might be reduced by another 6% through the implementation of other pest-control techniques, including cultivation of short-season cotton, fertilizer and water management, improved sanitation, crop rotations, the use of crop seed cleaned of weed seeds during culture and processing, and altered tillage practices (Grimes 1985, OTA 1979). Depending on the particular environment, insecticide use on cotton might even be reduced much more. For example, Shaunak et al. (1982) reported that insecticide use in the Lower Rio Grande Valley of Texas could be reduced 97% by planting short-season cotton under dryland conditions. This practice also resulted in a twofold increase in net profits over conventional methods.

Thus, by implementing combinations of monitoring, application technology, short-season cotton, fertilizer and water management, improved sanitation, crop rotations, and tillage practices in cotton, insecticide use might be reduced 40%. These alternative controls should pay for themselves through reduced insecticide and application costs.

Table 3. Current and potential reduced use of pesticides in US crops. Total current pesticide costs plus total added alternative control costs (Pimentel et al. 1991).

	Pesticide use (million kg)		Cost (million dollars)	
	Current	Potential reduced	Total	Added alternative control
Insecticides	62.1	27.2	$ 817.8	156.5
Herbicides	219.6	98.9	3115.8	845.3
Fungicides	37.6	26.6	207.4	16.5
Total	319.3	152.7	4141.0	1018.3

Herbicides

Corn and soybeans account for approximately 70% of the total herbicide applied in agriculture (Pimentel and Levitan 1986). We use these crops to illustrate the potential of decreasing herbicide use.

Corn. More than half (53%) of the herbicides used on crops are applied to corn (Pimentel and Levitan 1986). More than 3 kg of herbicide are applied per hectare of corn, and more than 90% of the corn hectarage planted is treated. By not requiring total elimination of weeds, in some cases herbicide use can be reduced by 75% (Schweizer 1989). At present, 91% of the corn land is also culti-

[4]R. B. Metcalf, 1990, personal communication. University of Illinois, Champaign.

Table 4. Total estimated environmental and social costs for pesticides in the United States. (Modified and updated from Pimentel et al. 1980.)

Environmental factor	Cost ($ millions)
Human pesticide poisonings	250
Animal pesticide poisonings and contaminated livestock products	15
Reduced natural enemies	150
Pesticide resistance	150
Honeybee poisonings and reduced pollination	150
Losses of crops and trees	75
Fishery and wildlife losses	15
Government pesticide pollution regulations	150
Monitoring wells and groundwater	1200
Total	2155

vated to help control weeds (Duffy 1982).

The average costs and returns per hectare to no-till, reduced-till, and conventional-till culture have actually been found to be quite similar (Duffy and Hanthorn 1984). For example, added labor, fuel, and machinery costs for conventional-till practices for corn were approximately $24/ha higher than those for no-till. However, the costs for the added fertilizers, pesticides, and seeds in the no-till system were $22/ha higher than conventional-till (Duffy and Hanthorn 1984).

It might be possible to reduce herbicide use on corn by approximately 60% if the use of mechanical cultivation and rotations were increased (Forcella and Lindstrom 1988). Corn and soybean rotations have been found to provide substantially higher returns than either crop grown separately and continuously (Helmers et al. 1986). However, we estimate that weed control costs will increase by approximately 30%, because not all alternative practices and rotations are profitable.

Soybeans. The second-largest amount of herbicides is applied to soybeans, with approximately 96% of soybean hectarage receiving treatment for weed control; 96% of the hectarage also receives some tillage and mechanical cultivation for weed control (Duffy 1982). Several techniques have been developed that increase the efficiency of chemical applications. The rope-wick applicator has been used in soybeans to reduce herbicide use approximately 90%, and this applicator was found to increase soybean yields 51% over conventional treatments (Dale 1980). Also, a new model of recirculating sprayer saves 70–90% of the spray emitted that is not trapped by the weeds (Matthews 1985). Spot treatments are a third method of decreasing unnecessary pesticide applications.

In addition, alternative techniques are available to reduce the need for herbicide use on soybeans. These include ridge-till, tillage, mechanical cultivation, row spacing, planting date, tolerant varieties, crop rotations, spot treatments, and reduced dosages (Forcella and Lindstrom 1988, Helmers et al. 1986, Russnogle and Smith 1988, Tew et al. 1982). Employing several of these alternative techniques in combination might reduce herbicide use in soybeans by approximately 60%. Despite the results of Tew et al. (1982) that indicate no added control costs for the alternatives, we estimate that the costs of weed control would increase $10/ha (Pimentel et al. 1991).

Fungicides

In considering the possible reduction of fungicide use, apples and potatoes were selected as examples. These two crops account for 26% of all the fungicides used in agriculture (Pimentel et al. 1991).

Apples. Most fungicides are applied to apples, peaches, citrus, and other fruit crops (Pimentel and Levitan 1986). Integrated pest management data from apples in New York State suggest that fungicide use on apples could be reduced approximately 10% by monitoring and better forecasting of disease based on weather data (Kovach and Tette 1988).

In addition, a recent design in spray nozzle and application equipment demonstrated that the amount of fungicide applied for apple scab control could be reduced by 50% (Van der Scheer 1984). Thus, by employing better weather forecasting and improved application technology combined with scouting, fungicide use on apples could be reduced an estimated 20% (Pimentel et al. 1991).

Potatoes. Approximately 96% of potato hectarage is treated with fungicides (Pimentel et al. 1991). Without fungicide treatments, losses from diseases ranged from 5% to 25% depending on the year, weather, and pathogen spread, whereas losses with fungicide treatments were reported to be approximately 20% in a controlled experiment (Teng and Bissonnette 1985). Shields et al. (1984) reported that the planting of short-season potatoes in Wisconsin reduced the number of required fungicide applications by one-third.

Correct storage, handling, and planting of seed tubers and proper management of soil moisture and fertility minimize losses to most diseases (Rich 1991). Forecasting and monitoring might also be employed to reduce fungicide use 15–25% (Royle and Shaw 1988). Monitoring should concentrate on disease incidence and forecasting weather conditions so fungicides can be applied before infection outbreak. Employing a combination of these controls, it might be possible to reduce fungicide use on potatoes approximately one-third, at an estimated cost of $5/ha (Pimentel et al. 1991).

Overall pesticide-reduction assessment

By substituting nonchemical alternatives for some pesticides used on 40 major crops, we estimate that total agricultural pesticide use can be reduced by approximately 50% (Pimentel et al. 1991). The added costs for implementing these alternatives are estimated to be approximately $1 billion (Table 3). These alternatives would increase total pest-control costs approximately 25% and would increase total food production costs at the farm 0.6%.

If alternative technologies that result in reduced crop yields became acceptable, the assessment of benefit and cost relationships would be quite different. For example, each 1.0% decrease in crop yield in agriculture results in a corresponding 4.5% increase in the farm price of goods.[5] It is also important to note that over-

[5]D. Sisler, 1988, personal communication. Cornell University, Ithaca, NY.

production is the prime reason that the United States spends $26 billion annually on price supports (USOMB 1989). Thus, if changes in pest control did in fact reduce yields, it might both increase farm income and decrease government subsidy expenses.

Environmental and health aspects

Society now pays a high price for its use of pesticides. Pesticide-control measures cost approximately $4.1 billion annually, not including the indirect environmental and public-health costs, which total more than $2.2 billion annually (Pimentel et al. 1991).

Perhaps the most serious social and environmental costs related to pesticide use are the human pesticide poisonings. Annually approximately 20,000 accidental poisonings occur, mostly from agricultural pesticides, with 2000 cases requiring hospitalization.[6] These poisonings result in approximately 50 fatalities per year. Pesticides are also implicated in numerous other human diseases, including cancer and sterility. An estimated 6000 human cases of pesticide-induced cancer occur each year (EPA 1987).

The costs of human poisonings and other detrimental environmental effects of pesticide use are delineated in Table 4:

- A large number of domestic animals are poisoned each year by pesticides. Significant amounts of meat and milk are contaminated with pesticides and must be destroyed.
- When pesticides are applied to crops, natural enemies important for controlling some pests are frequently destroyed (OTA 1979). This destruction causes pest outbreaks that must be controlled with additional pesticide applications.
- The development of pesticide resistance in pest populations is another major environmental problem. This resistance requires additional pesticide treatments and more costly controls.
- Large numbers of honeybees and wild bees are poisoned by pesticides, resulting in honey losses and reduced crop pollination.
- Some pesticides, especially when applied by aircraft, drift onto adjacent agricultural lands and damage or destroy crops and forest resources.
- Significant numbers of fish and other wildlife are killed by pesticides each year.
- The government program that attempts to regulate and limit pesticide pollution also incurs major administrative costs (Pimentel et al. 1991).
- Monitoring wells and groundwater for pesticide contamination is costly (Nielsen and Lee 1987).

This analysis is an incomplete assessment of the existing environmental problems caused by pesticide use. There is no completely satisfactory way to summarize all of the environmental and social costs or benefits in terms of dollars. For example, it is impossible to place a monetary value on human death, disease, or disability.

The nearly $1 billion attributed to environmental and social costs (Pimentel et al. 1980) represents only a small portion of the actual costs. A more complete accounting of the indirect costs would also include unrecorded losses of fish, wildlife, crops, and trees; losses resulting from the destruction of soil invertebrates, microflora, and microfauna; even higher costs of human poisonings; chronic health problems such as cancer; groundwater contamination; and contamination of human food other than livestock products (Blume 1987). If the full environmental and social costs could be estimated, the total cost could rise to as much as $4 billion annually.

Although the nonchemical alternative controls proposed as substitutes for pesticides in this study are significantly safer than pesticides, the alternatives themselves may cause some social and environmental problems (Pimentel et al. 1984). However, if one assumes that reducing pesticide use by 50% might also eventually reduce the environmental and public health risks from pesticides by approximately one-half, then the added costs for the nonchemical alternatives ($1 billion) might be offset by the reduced environmental and public health risks.

Conclusions

Pesticides cause serious public health problems and considerable damage to agricultural and natural ecosystems. A conservative estimate suggests that the environmental and social costs of pesticide use in the United States are at least $2.2 billion annually, and the actual cost is probably double this amount. In addition to these costs, the nation spends $4.1 billion annually to treat crops with 320 million kg of pesticides.

This article confirms previous reports that it is feasible to reduce pesticide use by one-half. The cost is estimated at $1 billion. Such a finding supports the projections of the Office of Technology Assessment (1979) and the National Academy of Sciences (1989), as well as the policies adopted by the Danish and Swedish governments intended to reduce pesticide use 50%.

The 50% pesticide reduction in our current assessment would help satisfy the concerns of the majority of the public, who worry about pesticide levels in their food as well as damage to the environment (Sachs et al. 1987). If pesticide use were reduced by one-half without any decline in crop yield, the total price increase in purchased food, due to increased costs of alternative controls, is calculated to be only 0.6%. If assured that pesticides in food and the environment were greatly reduced, it is likely that the public would be willing to pay this slight increase in food costs.

Higher cosmetic standards have resulted in greater quantities of pesticides being applied to food crops. The rapidly growing use of pesticides for cosmetic purposes is detrimental to both public health and the environment, and it is also contrary to public demand (Pimentel et al. 1977). The public would accept some reduction in cosmetic standards if it would result in a reduction in pesticide contamination of food (Healy 1989). This acceptance is indicated by the growing popularity of organic food

[6] J. Blondell, 1989, personal communication. US Environmental Protection Agency, Washington, DC.

stores and supermarkets that guarantee pesticide-free foods (Hammit 1986, Poe 1988). Furthermore, the presence of parts of soft-bodied insects in highly processed foods, such as catsup and applesauce, carries no risk to public health and even is of some nutritional value (Pimentel et al. 1977).

To estimate more accurately the potential of reduced pesticide use, more data are needed, especially concerning those agricultural technologies that have contributed to the increase in pesticide use during the past 40 years while simultaneously increasing crop losses to pests. We hope that more complete data will be assembled so that more detailed analyses can be made.

Implementing a program to reduce pesticide use in agriculture will require the combined education of farmers and the public and some new regulations. In addition, it will require that the federal government revise current policies, like its commodity and price-support program, that prevent farmers from employing crop rotations and other sound agricultural practices (NAS 1989). Several current government policies actually increase pest problems and pesticide use (NAS 1989).

At the same time, a greater investment is needed in research on alternative pest-control practices. Many opportunities exist to reduce pesticides through the implementation of new environmental, cultural, and biological pest controls (Pimentel 1991). We strongly support the National Academy of Sciences' research recommendations for alternative pest controls (NAS 1989).

If the public is concerned about pesticides contaminating their food and environment, do the small economic costs of reducing pesticide use outweigh the ecological and public health benefits? We hope that the public and state and federal governments will investigate the ecology, economics, and ethics of pesticide reduction in agriculture. A careful assessment must be made to evaluate the benefits and risks of pesticides and nonchemical alternatives for society.

Acknowledgments

We thank the following people for reading an earlier draft of this article, for their many helpful suggestions, and, in some cases, for providing additional information: B. Barclay and M. El-Ashry, World Resources Institute; R. Frans, University of Arkansas; J. Hatchett, Kansas State University; M. Harris, Texas A&M University; H. Hokkanen, Agricultural Research Centre, Jokioinen, Finland; D. Horn, Ohio State University; C. Huffaker, University of California, Berkeley; H. Janzen, Agriculture Canada; P. Johnson, University of New Hampshire; F. McEwen, University of Guelph; B. Mogensen, National Environmental Research Institute, Denmark; I. Oka, Bogor Research Institute for Field Crops, Indonesia; C. Osteen, USDA, Washington, DC; M. Pathak, International Rice Research Institute, Philippines; J. Pierce, Environmental Action, Washington, DC; E. Radcliffe, University of Minnesota; J. Schalk, USDA Vegetable Lab, Charleston, NC; D. Wen, Chinese Academy of Sciences; K. Stoner, Connecticut Agricultural Experiment Station; G. Surgeoner, University of Guelph; and G. Teetes, Texas A&M University. We also thank T. Dennehy, E. Glass, R. Roush, J. Tette, and C. Wien at Cornell University. We gratefully acknowledge the partial support of this study by a William and Flora Hewlett Foundation Grant to the Centre for Environmental Research, Cornell University.

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Environmental and Economic Costs of Pesticide Use

An assessment based on currently available US data, although incomplete, tallies $8 billion in annual costs

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Worldwide, approximately 2.5 million tons of pesticides are applied each year with a purchase price of $20 billion (Pesticide News 1990). In the United States, approximately 500,000 tons of 600 different types of pesticides are used annually at a cost of $4.1 billion, including application costs (Pimentel et al. 1991).

Pesticides make a significant contribution to maintaining world food production. In general, each dollar invested in pesticide control returns approximately $4 in crops saved. Estimates are that losses to pests would increase 10% if no pesticides were used at all; specific crop losses would range from zero to nearly 100%.

Despite the widespread use of pesticides in the United States, pests (principally insects, plant pathogens, and weeds) destroy 37% of all potential food and fiber crops (Pimentel 1990). Although pesticides are generally profitable, their use does not always decrease crop losses. For example, even with the tenfold increase in insecticide use in the United States from 1945 to 1989, total crop losses from insect damage have nearly doubled from 7% to 13% (Pimentel et al. 1991). This rise in crop losses to insects is, in part, caused by changes in agricultural practices. For instance, the replacement of rotating corn with other crops with the continuous production on approximately half the hectarage has resulted in nearly a fourfold increase in corn losses to insects, despite a thousandfold increase in insecticide use in corn production (Pimentel et al. 1991).

Indirect costs must be examined to facilitate a balanced, sound policy of pesticide use

Most benefits of pesticides are based only on direct crop returns. Such assessments do not include the indirect environmental and economic costs associated with pesticides. To facilitate the development and implementation of a balanced, sound policy of pesticide use, these costs must be examined. More than a decade ago, the US Environmental Protection Agency (EPA) pointed out the need for such a risk investigation (EPA 1977). So far only a few papers on this difficult subject have been published.

The obvious need for an updated and comprehensive study prompted our investigation of the complex of environmental and economic costs resulting from the nation's dependence on pesticides. Included in the assessment are analyses of pesticide impacts such as human health effects; domestic animal poisonings; increased control expenses resulting from pesticide-related destruction of natural enemies and from the development of pesticide resistance; crop pollination problems and honeybee losses; crop and crop product losses; groundwater and surface water contamination; fish, wildlife, and microorganism losses; and governmental expenditures to reduce the environmental and social costs of pesticide use.

Human health effects

Human pesticide poisonings and illnesses are clearly the highest price paid for pesticide use. A recent World Health Organization and United Nations Environmental Programme report (WHO/UNEP 1989) estimated there are 1 million human pesticide poisonings each year in the world, with approximately 20,000 deaths. In the United States, nonfatal pesticide poisonings reported by the American Association of Poison Control Centers total approximately 67,000 each year (Litovitz et al. 1990). J. Blondell[1] has indicated that because of demographic gaps, this figure represents only 73% of the total. According to Blondell, the number of accidental (no suicide or homicide) fatalities is approximately 27 per year.

Although developed countries, including the United States, annually use approximately 80% of all the pesticides produced in the world (Pimentel 1990), less than half of the pesticide-induced deaths occur in these countries (House of Commons Agri-

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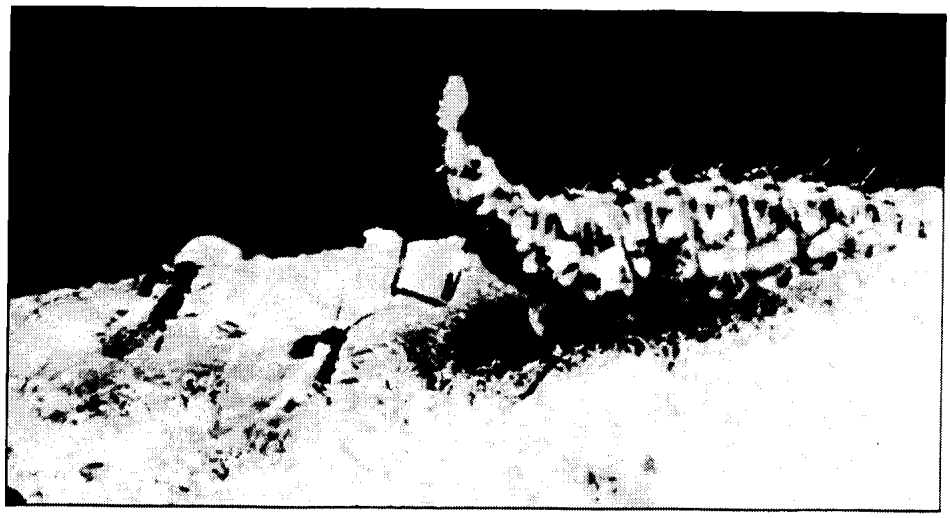
[1]J. Blondell, 1990, personal communication. Environmental Protection Agency, Washington, DC.

culture Committee 1987). A higher proportion of pesticide poisonings and deaths occurs in developing countries where there are inadequate occupational and other safety standards, insufficient enforcement, poor labeling of pesticides, illiteracy, inadequate protective clothing and washing facilities, and insufficient knowledge of pesticide hazards by users.

Both the acute and chronic health effects of pesticides warrant concern. The acute toxicity of most pesticides is well documented (Ecobichon et al. 1990), but information on chronic human illnesses resulting from pesticide exposure, including cancer, is weak. The International Agency for Research on Cancer found "sufficient" evidence of carcinogenicity for 18 pesticides and "limited" evidence of carcinogenicity for an additional 16 pesticides based on animal studies (WHO/UNEP 1989).

With humans, the evidence concerning cancer is also mixed. For example, a recent study in Saskatchewan indicated no significant difference in non-Hodgkin's lymphoma mortality between farmers and nonfarmers (Wigle et al. 1990), whereas other studies have reported some cancer in farmers (WHO/UNEP 1989). It is estimated that the number of US cases of cancer associated with pesticides in humans is less than 1% of the nation's total cancer cases.[2] Considering that there are approximately 1 million cancer cases per year (USBC 1990), Schottenfeld's assessment suggests that less than 10,000 cases of cancer are due to pesticides per year.

Many other acute and chronic maladies are beginning to be associated with pesticide use. For example, the recently banned pesticide, used for plant pathogen control, dibromochloropropane (DBCP) caused testicular dysfunction in animal studies (Foote et al. 1986) and was linked with infertility among human workers exposed to DBCP (Potashnik and Yanai-Inbar 1987). Also, a large body of evidence has been accumulated over recent years from animal studies suggesting pesticides can produce immune dysfunction (Thomas and House 1989). In a study of women who had chronically ingested groundwater contaminated with low levels of aldicarb (used for insect control; mean 16.6 ppb), Fiore et al. (1986) reported evidence of significantly reduced immune response, although these women did not exhibit any overt health problems.



Aphid lions can be purchased to fight insect pests. This late-stage larva (right), approximately 10 mm long, preys on aphids (left) and other small soft-bodied insects. With its hollow mandibles, the predator pierces its prey to suck out the blood. Photo: USDA.

Of particular concern are the chronic health problems associated with effects of organophosphorus pesticides, which have largely replaced the banned organochlorines (Ecobichon et al. 1990). The malady organophosphate-induced delayed polyneuropathy is well documented and includes irreversible neurological defects (Lotti 1984). Other defects in memory, mood, and abstraction have been documented. The evidence confirms that persistent neurotoxic effects may be present even after the termination of an acute poisoning incident (Ecobichon et al. 1990, Rosenstock et al. 1991).

Such chronic health problems are a public health issue, because everyone, everywhere is exposed to some pesticide residues in food, water, and the atmosphere. Fruits and vegetables receive the highest dosages of pesticides. Approximately 35% of the foods purchased by US consumers have detectable levels of pesticide residues (FDA 1990). From 1% to 3% of the foods have pesticide residue levels above the legal tolerance level (FDA 1990, Hundley et al. 1988). These residue levels could well be higher because the US analytical methods now employed detect only approximately one-third of the more than 600 pesticides in use (OTA 1988). Therefore, there are many reasons why 97% of the public is genuinely concerned about pesticide residues in their food (FDA 1989).

Medical specialists are concerned about the lack of public health data about pesticide effects in the United States (GAO 1986). Based on an investigation of 92 pesticides used on food, GAO (1986) estimates data on health problems associated with registered pesticides contains little or no information on tumors and birth defects.

Although no one can place a precise monetary value on a human life, studies done for the insurance industry have computed monetary ranges for the value of a "statistical life" between $1.6 and $8.5 million (Fisher et al. 1989). For our assessment, we use the conservative estimate of $2 million per human life. Based on the available data, estimates are that human pesticide poisonings and related illnesses in the United States total approximately $787 million each year (Table 1).

Animal poisonings and contaminated products

In addition to pesticide problems that affect humans, several thousand domestic animals are poisoned by pesticides each year; meat, milk, and eggs

[2]D. Schottenfeld, 1991, personal communication. College of Medicine, University of Michigan, Ann Arbor.

Table 1. Estimated economic costs of human pesticide poisonings and other pesticide-related illnesses in the United States each year.

Effects	Cost ($ million/year)
Hospitalization after poisonings: 2380* × 2.84 days @ $1000/day	6.759
Outpatient treatment after poisonings: 27,000† × $630‡	17.010
Lost work due to poisonings: 4680* workers × 4.7 days × $80/day	1.760
Treatment of pesticide-induced cancers: <10,000§ cases × $70,700†/case	707.000
Fatalities: 27 accidental fatalities† × $2 million	54.000
Total	786.529

*Keefe et al. 1990.
†J. Blondell, 1991, personal communication. EPA, Washington, DC.
‡Includes hospitalization, foregone earnings, and transportation (Castillo and Appel 1989).
§See text for details.

are also contaminated. Of 25,000 calls made to the Illinois Animal Poison Control Center in 1987, nearly 40% concerned pesticide poisonings in dogs and cats (Beasley and Trammel 1989). Similarly, Kansas State University reported that 67% of all animal pesticide poisonings involve dogs and cats (Barton and Oehme 1981). This large representation is not surprising, because dogs and cats usually wander freely about the home and farm and therefore have greater opportunity to come into contact with pesticides than other domesticated animals.

The best estimates indicate that approximately 20% of the total monetary value of animal production, or approximately $4.2 billion, is lost to all animal illnesses, including pesticide poisonings (Pimentel et al. in press). Colvin (1987) reported that 0.5% of animal illnesses and 0.04% of all animal deaths reported to a veterinary diagnostic laboratory were due to pesticide toxicosis. Thus, $30 million in domestic animals are lost to pesticide poisonings (Pimentel et al. in press).

This estimate is based only on poisonings reported to veterinarians. Many animal pesticide poisonings that occur in the home and on farms go undiagnosed and are attributed to other factors. In addition, when a farm animal poisoning occurs and little can be done for an animal, the farmer seldom calls a veterinarian but either waits for the animal to recover or destroys the animal.[3]

Additional economic losses occur when meat, milk, and eggs are contaminated with pesticides. In the United States, all animals slaughtered for human consumption, if shipped interstate, and all imported meat and poultry must be inspected by the US Department of Agriculture. This inspection is to ensure that the meat and products are wholesome, properly labeled, and do not present a health hazard. One part of this inspection, which involves monitoring meat for pesticide and other chemical residues, is the responsibility of the National Residue Program.

Of more than 600 pesticides now in use, National Residue Program tests are made for only 41,[4] which have been determined by the Federal Drug Administration, the Environmental Protection Agency, and Food Safety and Inspection Service to be of public health concern. Although the monitoring program records the number and type of violations, there is no significant cost to the animal industry because the meat is generally sold and consumed before the test results are available. Approximately 3% of the chickens with illegal pesticide residues are sold in the market (NAS 1987).

When the costs attributable to domestic animal poisonings and contaminated meat, milk, and eggs are combined, the economic value of all livestock products in the United States lost to pesticide contamination is estimated to be at least $29.6 million annually. Similarly, other nations lose significant numbers of livestock and large amounts of animal products each year due to pesticide-induced illness or death. Exact data concerning these livestock losses do not exist, and the available information comes only from reports of the incidence of mass destruction of livestock. For example, when the pesticide leptophos was used by Egyptian farmers on rice and other crops, 1300 draft animals were poisoned and lost.[5]

Destruction of beneficial natural predators and parasites

In both natural and agricultural ecosystems, many species, especially predators and parasites, control or help control herbivorous populations. Indeed, these natural beneficial species make it possible for ecosystems to remain foliated. With parasites and predators keeping herbivore populations at low levels, only a relatively small amount of plant biomass is removed each growing season (Hairston et al. 1960). Natural enemies play a major role in keeping populations of many insect and mite pests under control (DeBach 1964).

Like pest populations, beneficial natural enemies are adversely affected by pesticides (Croft 1990). For example, pests have reached outbreak levels in cotton and apple crops following the destruction of natural enemies by pesticides. Among such cotton pests are cotton bollworm, tobacco budworm, cotton aphid, spider mites, and cotton looper (OTA 1979). The apple pests in this category include European red mite, red-banded leafroller, San Jose scale, oystershell scale, rosy apple aphid, woolly apple aphid, white apple leafhopper, two-spotted spider mite, and apple rust mite (Croft 1990). Significant pest outbreaks also have occurred in other crops (Croft 1990, OTA 1979). Because parasitic and predacious insects often have complex searching and attack behaviors, sublethal insecticide dosages may alter this behavior and in this way disrupt effective biological controls.[6]

Fungicides also can contribute to pest outbreaks when they reduce fungal pathogens that are naturally parasitic on many insects. For example, the use of benomyl, used for plant pathogen control, reduces populations of entomopathogenic fungi. This effect results in increased survival of

[3]G. Maylin, 1977, personal communication. College of Veterinary Medicine, Cornell University, Ithaca, NY.
[4]D. Beerman, 1991, personal communication. Department of Animal Science, Cornell University, Ithaca, NY.
[5]A. H. El Sebae, 1992, personal communication. University of Alexandria, Alexandria, Egypt.
[6]L. E. Ehler, 1991, personal communication. University of California, Davis.

velvet bean caterpillars and cabbage loopers in soybeans. The increased number of insects eventually leads to reduced soybean yields (Johnson et al. 1976).

When outbreaks of secondary pests occur because their natural enemies are destroyed by pesticides, additional and sometimes more expensive pesticide treatments have to be made in efforts to sustain crop yields. This consequence raises overall costs and contributes to pesticide-related problems. An estimated $520 million can be attributed to costs of additional pesticide applications and increased crop losses, both of which follow the destruction of natural enemies by pesticides applied to crops (Pimentel et al. in press).

Worldwide, as in the United States, natural enemies are being adversely affected by pesticides. Although no reliable estimate is available concerning the impact of the loss in terms of increased pesticide use and/or reduced yields, general observations by entomologists indicate that the impact of loss of natural enemies is severe in many parts of the world. For example, from 1980 to 1985, insecticide use in rice production in Indonesia drastically increased (Oka 1991). This usage caused the destruction of beneficial natural enemies of the brown planthopper, and the pest populations exploded. Rice yields dropped to the extent that rice had to be imported into Indonesia for the first time in many years. The estimated loss in rice in just a two-year period was $1.5 billion (FAO 1988).

After that incident, entomologist I. N. Oka and his cooperators, who previously had developed a successful low-insecticide program for rice pests in Indonesia, were consulted by Indonesian President Soeharto's staff.[7] Their advice was to substantially reduce insecticide use and return to a sound treat-when-necessary program that protected the natural enemies. Following Oka's advice, President Soeharto mandated in 1986 that 57 of 64 pesticides would be withdrawn from use on rice and pest management practices would be improved. Pesticide subsidies to farmers also were eliminated. Subsequently, rice yields increased to levels well above those recorded during the period of heavy pesticide use (FAO 1988).

Biocontrol specialist D. Rosen[8] estimates that natural enemies account for up to 90% of the control of pest species achieved in agroecosystems and natural systems; we estimate that about half of the control of pest species is due to natural enemies. Pesticides give an additional control of 10%, and the remaining percentage is due to host-plant resistance and other limiting factors present in the agroecosystem.

Pesticide resistance in pests

In addition to destroying natural enemy populations, the extensive use of pesticides has often resulted in the development of pesticide resistance in insect pests, plant pathogens, and weeds. In a report of the United Nations Environment Programme, pesticide resistance was ranked as one of the top four environmental problems in the world (UNEP 1979). Approximately 504 insect and mite species (Georghiou 1990), a total of nearly 150 plant pathogen species, and about 273 weed species are now resistant to pesticides (Pimentel et al. in press).

Increased pesticide resistance in pest populations frequently results in the need for several additional applications of the commonly used and different pesticides to maintain expected crop yields. These additional pesticide applications compound the problem by increasing environmental selection for resistance traits. Despite attempts to deal with it, pesticide resistance continues to develop (Dennehy et al. 1987).

The impact of pesticide resistance, which develops gradually over time, is felt in the economics of agricultural production. A striking example of such development occurred in northeastern Mexico and the Lower Rio Grande of Texas (Adkisson 1972). Extremely high pesticide resistance had developed in the tobacco budworm population on cotton. Finally, in early 1970, approximately 285,000 ha of cotton had to be abandoned because pesticides were ineffective and there was no way to protect the crop from the budworm. The economic and social impacts on these Texan and Mexican farming communities that depend on cotton were devastating.

Cotton pests, such as this cotton boll worm, have reached outbreak levels after pesticides destroyed their natural enemies.

A study by Carrasco-Tauber (1989) indicates the extent of costs attributed to pesticide resistance. This study reported a yearly loss of $45 to $120/ha to pesticide resistance in California cotton. A total of 4.2 million hectares of cotton were harvested in 1984, thus assuming a loss of $82.50/ha; therefore, approximately $348 million of California cotton crop was lost to resistance. Because $3.6 billion of US cotton were harvested in 1984, the loss due to resistance for that year was approximately 10%. Assuming a 10% loss in other major crops that receive heavy pesticide treatments in the United States, crop losses due to pesticide resistance are estimated to be $1.4 billion/yr.

A detailed study by Archibald (1984) further demonstrated the hidden costs of pesticide resistance in California cotton. She reported that 74% more organophosphorus insecticides were required in 1981 to achieve the same kill of pests, like *Heliothis* spp. (cotton bollworm and budworm), than in 1979. Her analysis demonstrated that the diminishing effect of pesticides plus intensified pest control reduced the economic return per dollar of pesticide invested to only $1.14.

Furthermore, efforts to control resistant *Heliothis* spp. exact a cost on other crops when large, uncontrolled

[7] I. N. Oka, 1990, personal communication. Bogor Research Institute for Food Crops, Bogor, Indonesia.

[8] D. Rosen, 1991, personal communication. Hebrew University of Jerusalem, Jerusalem, Israel.

populations of *Heliothis* and other pests disperse onto other crops. In addition, the cotton aphid and the whitefly exploded as secondary cotton pests because of their resistance and their natural enemies' exposure to the high concentrations of insecticides.

The total external cost attributed to the development of pesticide resistance is estimated to range between 10% and 25% of current pesticide treatment costs (Harper and Zilberman 1990), or approximately $400 million each year in the United States alone. In other words, at least 10% of pesticide used in the United States is applied just to combat increased resistance that has developed in various pest species.

In addition to plant pests, a large number of insect and mite pests of both livestock and humans have become resistant to pesticides. Although a relatively small quantity of pesticide is applied for control of pests of livestock and humans, the cost of resistance has become significant. Based on available data, we estimate the yearly cost of resistance in such pests to be approximately $30 million for the United States.

Although the costs of pesticide resistance are high in the United States, its costs in tropical developing countries are significantly greater, because pesticides are used there not only to control agricultural pests but also for the control of disease vectors.

One of the major costs of resistance in tropical countries is associated with malaria control. By 1961, the incidence of malaria in India after early pesticide use had declined from several million cases to only 41,000 cases. However, because mosquitoes developed resistance to pesticides and malarial parasites developed resistance to drugs, the incidence of malaria in India now has exploded to approximately 59 million cases per year (NAS 1991). Similar problems are occurring in the rest of Asia, Africa, and South America, with the total incidence of malaria estimated to be 270 million cases (NAS 1991).

Bee poisonings and reduced pollination

Honeybees and wild bees are vital for pollination of crops including fruits and vegetables. Their direct and indirect benefits to agricultural production range from $10 billion to $33 billion each year in the United States (Robinson et al. 1989).[9] Because most insecticides used in agriculture are toxic to bees, pesticides have a major impact on both honeybee and wild bee populations. D. Mayer[10] estimates that 20% of all losses of honeybee colonies are due to pesticide exposure; this includes colonies that are killed outright or die during the winter. Mayer calculates that the direct annual loss reaches $13.3 million (Table 2). Another 15% of the bee colonies either are seriously weakened by pesticides or suffer losses when apiculturists have to move colonies to avoid pesticide damage.

According to Mayer, the yearly estimated loss from partial bee kills, reduced honey production, plus the cost of moving colonies totals approximately $25 million. Also, as a result of heavy pesticide use on certain crops, beekeepers are excluded from 4 to 6 million hectares of otherwise suitable apiary locations.[11] Mayer estimates the yearly loss in potential honey production in these regions is approximately $27 million.

In addition to these direct losses caused by damage to bees and honey production, many crops are lost because of the lack of pollination. In California, for example, approximately 1 million colonies of honey bees are rented annually at $20 per colony to augment the natural pollination of almonds, alfalfa, melons, and other fruits and vegetables.[12] Because California produces nearly 50% of US bee-pollinated crops, the total cost for bee rental for the entire country is estimated at $40 million. Of this cost, we estimate at least one-tenth or $4 million is attributed to the effects of pesticides (Table 2).

Estimates of annual agricultural losses due to the reduction in insect pollination of crops by pesticides may range as high as $4 billion per year.[13] For most crops, both crop yield and quality are enhanced by effective pollination. For example, McGregor et al. (1955) demonstrated that for several cotton varieties, effective pollination by bees resulted in yield increases from 20% to 30%. Assuming that a conservative 10% increase in cotton yield would result from more efficient pollination and subtracting charges for bee rental, the net annual gain for cotton alone could be as high as $400 million. However, using bees to enhance cotton pollination is currently impossible because of the intensive use of insecticides on cotton.

Table 2. Estimated honeybee losses and pollination losses from honeybees and wild bees.

Loss	Cost ($ million/year)
Colony losses from pesticides	13.3
Honey and wax losses	25.3
Loss of potential honey production	27.0
Bee rental for pollination	4.0
Pollination losses	200.0
Total	319.6

Mussen (1990) emphasizes that poor pollination not only reduces crop yields, but, more important, it reduces the quality of crops, especially fruit such as melons. In experiments with melons, E. L. Atkins[14] reported that with adequate pollination melon yields were increased 10% and quality was raised 25% as measured by the dollar value of the crop.

Based on the analysis of honeybee and related pollination losses caused by pesticides, pollination losses attributed to pesticides are estimated to represent approximately 10% of pollinated crops and have a yearly cost of approximately $200 million. Adding these costs to the other environmental costs of pesticides on honeybees and wild bees, the total annual loss is calculated to be approximately $320 million (Table 2). Therefore, the available evidence confirms that the yearly cost of direct honeybee losses, together with reduced yields resulting from poor pollination, are significant.

[9]E.L. Atkins, 1990, personal communication. University of California, Riverside.
[10]D. Mayer, 1990, personal communication. Department of Entomology, Washington State University, Pullman.
[11]See footnote 10.
[12]R. A. Morse, 1990, personal communication. Department of Entomology, Cornell University, Ithaca, NY.
[13]J. Lockwood, 1990, personal communication. Department of Entomology, University of Wyoming, Laramie.
[14]See footnote 9.

Crop and crop product losses

Basically, pesticides are applied to protect crops from pests in order to preserve yields, but sometimes the crops are damaged by pesticide treatments. This damage occurs when the recommended dosages suppress crop growth, development, and yield; pesticides drift from the targeted crop to damage adjacent nearby crops (e.g., citrus adjacent to cotton); residual herbicides either prevent chemical-sensitive crops from being planted in rotation or inhibit the growth of crops that are planted; and/or excessive pesticide residues accumulate on crops, necessitating the destruction of the harvest. Crop losses translate into financial losses for growers, distributors, wholesalers, transporters, retailers, and food processors. Potential profits as well as investments are lost. The costs of crop losses increase when the related costs of investigations, regulation, insurance, and litigation are added to the equation. Ultimately, the consumer pays for these losses in higher marketplace prices.

Data on crop losses due to pesticide use are difficult to obtain. Many losses are never reported to state and federal agencies because the injured parties often settle privately.[14,15] For example, in North Dakota, only an estimated one-third of the pesticide-induced crop losses are reported to the State Department of Agriculture.[16] Furthermore, according to the Federal Crop Insurance Corporation, losses due to pesticide use are not insurable because of the difficulty of determining pesticide damage.[17]

Damage to crops may occur even when recommended dosages of herbicides and insecticides are applied to crops under normal environmental conditions.[18] Recommended (heavy) dosages of insecticides used on crops have been reported to suppress growth and yield in both cotton and strawberry crops (ICAITI 1977). The increased susceptibility of some crops to insects and diseases after normal use of 2,4-D and other herbicides was demonstrated by Oka and Pimentel (1976). Furthermore, when weather and/or soil conditions are inappropriate for pesticide application, herbicide treatments may cause yield reductions ranging from 2% to 50% (Akins et al. 1976).

Crops are lost when pesticides drift from target crops to nontarget crops, sometimes located several miles downwind (Barnes et al. 1987). Drift occurs with almost all methods of pesticide application, including both ground and aerial equipment. The potential problem is greatest when pesticides are applied by aircraft; 50% to 75% of pesticides applied miss the target area (ICAITI 1977, Mazariegos 1985, Ware 1983). In contrast, 10% to 35% of the pesticide applied with ground-application equipment misses the target area (Hall 1991). The most serious drift problems are caused by speed sprayers and mist-blower sprayers, because with these application technologies approximately 35% of the pesticide drifts away from the target area. In addition, more of the total pesticide used in the US is applied with sprayers than with aircraft.[19]

Crop injury and subsequent loss due to drift is particularly common in areas planted with diverse crops. For example, in southwest Texas in 1983 and 1984, almost $20 million of cotton was destroyed from drifting 2,4-D herbicide when adjacent wheat fields were aerially sprayed with the herbicide (Hanner 1984).

When residues of some herbicides persist in the soil, crops planted in rotation may be injured (Keeling et al. 1989). In 1988/1989, an estimated $25 to $30 million of Iowa's soybean crop was lost due to the persistence of the herbicide Sceptor in the soil.[20]

Additional losses are incurred when food crops must be destroyed because they exceed the EPA regulatory tolerances for pesticide residue levels. Assuming that all the crops and crop products that exceed the EPA regulatory tolerances were destroyed as required by law, approximately $550 million in crops annually would be destroyed because of excessive pesticide contamination (Pimentel et al. in press). Because most of the crops with pesticides above the tolerance levels are neither detected nor destroyed, they are consumed by the public, avoiding financial loss to farmers but creating public health risks. In general, excess pesticides in the food go undetected unless a large number of people become ill after the food is consumed.

Table 3. Estimated loss of crops and trees due to the use of pesticides.

Impact	Cost ($ million/year)
Crop losses	136
Crop applicator insurance	245
Crops destroyed because of excess pesticide contamination	550
Investigations and testing	
Government	10
Private	1
Total	942

A well-publicized 1985 incident in California illustrates this problem. More than 1000 persons became ill from eating contaminated watermelons, and approximately $1.5 million dollars' worth of watermelons were ordered destroyed.[21] It was later learned that several California farmers treated watermelons with the insecticide aldicarb (Temik), which is not approved or registered for use on watermelons. After this crisis, the California State Assembly appropriated $6.2 million to be awarded to growers affected by state seizure and freeze orders (Legislative Counsel's Digest 1986). According to the California Department of Food and Agriculture, an estimated $800,000 in investigative costs and litigation fees resulted from this one incident.[22] The California Department of Health Services was assumed to have incurred similar expenses, putting the total cost of the incident at nearly $8 million.

Such costs as crop seizures and insurance should be added to the costs of direct crop losses due to the use of

[15]B. D. Berver, 1990, personal communication. Office of Agronomy Services, Brookings, SD.
[16]J. Peterson, 1990, personal communication. Pesticide/Noxious Weed Division, Department of Agriculture, Fargo, ND.
[17]E. Edgeton, 1990, personal communication. Federal Crop Insurance Corp., Washington, DC.
[18]J. Neal, 1990, personal communication. Chemical Pesticides Program, Cornell University, Ithaca, NY.

[19]See footnote 8.
[20]R. G. Hartzler, 1990, personal communication. Cooperative Extersion Service, Iowa State University, Ames.

[21]R. Magee, 1990, personal communication. California Department of Food and Agriculture, Sacramento.
[22]See footnote 21.

pesticides in commercial crop production. Then, the total monetary loss is estimated to be approximately $942 million annually in the United States (Table 3).

Groundwater and surface water contamination

Certain pesticides applied to crops eventually end up in groundwater and surface waters. The three most common pesticides found in groundwater are the insecticide aldicarb and the herbicides alachlor and atrazine (Osteen and Szmedra 1989). Estimates are that nearly one-half of the groundwater and well water in the United States is or has the potential to be contaminated (Holmes et al. 1988). EPA (1990a) reported that 10.4% of community wells and 4.2% of rural domestic wells have detectable levels of at least one pesticide of the 127 pesticides tested in a national survey. It would cost an estimated $1.3 billion annually in the United States to monitor well water and groundwater for pesticide residues (Nielsen and Lee 1987).

There are two major concerns about groundwater contamination with pesticides. First, approximately one-half of the population obtains its water from wells. Second, once groundwater is contaminated, the pesticide residues remain for long periods of time. Not only are there just a few microorganisms that have the potential to degrade pesticides, but the groundwater recharge rate averages less than 1% per year.

Monitoring pesticides in groundwater is only a portion of the total cost of US groundwater contamination. There is also the high cost of cleanup. For instance, at the Rocky Mountain Arsenal near Denver, Colorado, the removal of pesticides from groundwater and soil was estimated to cost approximately $2 billion (*New York Times* 1988). If all pesticide-contaminated groundwater were cleared of pesticides before human consumption, the cost would be approximately $500 million (based on the costs of cleaning water; Clark 1979). Note that the cleanup process requires a water survey to target the contaminated water for cleanup. Thus, adding monitoring and cleaning costs, the total cost of pesticide-polluted groundwater is estimated to be approximately $1.8 billion annually.

Fishery losses

Pesticides are washed into aquatic ecosystems by water runoff and soil erosion. Approximately 18 $t \cdot ha^{-1} \cdot yr^{-1}$ of soil are washed and/or blown from pesticide-treated cropland into adjacent locations, including streams and lakes (USDA 1989b). Pesticides also drift into streams and lakes and contaminate them (Clark 1989). Some soluble pesticides are easily leached into streams and lakes (Nielsen and Lee 1987).

Once in aquatic systems, pesticides cause fishery losses in several ways. High pesticide concentrations in water directly kill fish, low-level doses kill highly susceptible fish fry, and essential fish foods such as insects and other invertebrates are eliminated. In addition, because government safety restrictions ban the catching or sale of fish contaminated with pesticide residues, such unmarketable fish are considered an economic loss.

Each year, large numbers of fish are killed by pesticides. Based on EPA (1990b) data, we calculate that from 1977 to 1987 the cost of fish kills due to all factors has been 141 million fish/yr. Pesticides are the cause of 6–14 million of those deaths.

These estimates of fish kills are considered to be low. In 20% of the fish kills, no estimate is made of the number of fish killed. In addition, fish kills frequently cannot be investigated quickly enough to determine accurately the primary cause. Fast-moving waters in rivers dilute pollutants so that these causes of kills often cannot be identified. Moving waters also wash away some of the poisoned fish, whereas other poisoned fish sink to the bottom and cannot be counted. Perhaps most important, few if any of the widespread and more frequent low-level pesticide poisonings are dramatic enough to be observed. Therefore, most go unrecognized and unreported.

The average value of a fish has been estimated to be approximately $1.70, using the guidelines of the American Fisheries Society (AFS 1982); however, it was reported that Adolph Coors Company might be "fined up to $10 per dead fish, plus other penalties" for an accidental beer spill in a creek (*Barometer* 1991). At $1.70, the value of the low estimate of 6 to 14 million fish killed by pesticides per year is $10 to $24 million. The actual loss is probably several times this amount.

Wild birds

Wild birds are also damaged by pesticides; these animals make excellent indicator species. Deleterious effects on wildlife include death from direct exposure to pesticides or secondary poisonings from consuming contaminated prey; reduced survival, growth, and reproductive rates from exposure to sublethal dosages; and habitat reduction through elimination of food sources and refuges (McEwen and Stephenson 1979). In the United States, approximately 160 million ha/yr of land receives a heavy pesticide dose—averaging 3 kg per ha (Pimentel et al. 1991). With such a large area treated with heavy dosages, it is to be expected that the impact on wildlife is significant.

The full extent of bird and mammal destruction is difficult to determine because these animals are often secretive, camouflaged, highly mobile, and live in dense grass, shrubs, and trees. Typical field studies of the effects of pesticides often obtain extremely low estimates of bird and mammal mortality (Mineau and Collins 1988). Bird carcasses disappear quickly due to vertebrate and invertebrate scavengers, and field studies seldom account for birds that die a distance from the treated areas.

Nevertheless, many bird casualties caused by pesticides have been reported. For instance, White et al. (1982) reported that 1200 Canada geese were killed in one wheat field that was sprayed with a 2:1 mixture of parathion and methyl parathion at a rate of 0.8 kg/ha. Carbofuran applied to alfalfa killed more than 5000 ducks and geese in five incidents, whereas the same chemical applied to vegetable crops killed 1400 ducks in a single incident (Flickinger et al. 1991). Carbofuran is estimated to kill 1 to 2 million birds each year in the United States (EPA 1989). Another pesticide, diazinon, applied on just three golf courses, killed 700 Atlantic Brant geese or one-quarter of the wintering population of geese (Stone and Gradoni 1985).

Several studies report that the use of herbicides in crop production results in the elimination of weeds that harbor some insects (Potts 1986).[23] The use of herbicides has led to significant reductions in the gray partridge in the United Kingdom and the common pheasant in the United States. In the case of the partridge, population levels have decreased to less than 23% because partridge chicks (like pheasant chicks) depend on insects to supply them with protein needed for their development and survival (Potts 1986).[24]

Frequently, the form of a pesticide influences its toxicity to wildlife. For example, insecticide-treated seed and insecticide granules, including carbofuran, fensulfothion, fonofos, and phorate, are particularly toxic to birds when consumed. From 0.23 to 1.5 birds/ha are estimated to have been killed in Canada by these treated seed and granules, and in the United States estimates range from 0.25 to 8.9 birds/ha killed per year by the pesticides (Mineau 1988).

Pesticides also adversely affect the reproductive potential of many birds and mammals. Exposure of birds, especially predatory birds, to chlorinated insecticides has caused reproductive failure, sometimes attributed to eggshell thinning (Stickel et al. 1984). Most of the affected populations recovered after the ban of DDT in the United States. However, DDT and its metabolite DDE remain a concern; DDT continues to be used in developing countries, which contain wintering areas for numerous bird species (Stickel et al. 1984).

Although the gross values for wildlife are not available, expenditures are one measure of the monetary value. The money spent by bird hunters to harvest 5 million game birds was $1.1 billion, or approximately $216 per bird felled (USFWS 1988). It is estimated that approximately $0.40 per bird is spent for birdwatching (on travel and equipment), and $800 per bird is spent to rear and release a bird in the wild (Pimentel et al. in press). For our assessment, we place an average value per bird at $30.

If we assume that the damage pesticides inflict on birds occurs primarily on the 160 million ha of cropland that receives most of the pesticide, and the bird population is estimated to be 4.2 birds/ha of cropland (Blew 1990), then 672 million birds are directly exposed to pesticides. If it is conservatively estimated that only 10% of the bird population is killed, then the total number killed is 67 million birds. Note this estimate is at the lower end of the range of 0.25 to 8.9 birds/ha killed per year by pesticides mentioned earlier in this section. Also, this estimate is conservative because secondary losses to pesticide reductions in invertebrate-prey poisonings were not included in the assessment. Assuming the average value of a bird is $30, then an estimated $2 billion in birds are destroyed annually.

The US Fish and Wildlife Service spends $102 yearly on its Endangered Species Program, which aims to reestablish species, such as the bald eagle, peregrine falcon, osprey, and brown pelican, that in some cases were reduced by pesticides (USFWS 1991). Thus, when all the above costs are combined, we estimate that US bird losses associated with pesticide use represent a cost of approximately $2.1 billion/yr.

Table 4. Total estimated environmental and social costs from pesticides in the United States.

Impact	Cost ($ million/year)
Public health impacts	787
Domestic animal deaths and contamination	30
Loss of natural enemies	520
Cost of pesticide resistance	1400
Honeybee and pollination losses	320
Crop losses	942
Fishery losses	24
Bird losses	2100
Groundwater contamination	1800
Government regulations to prevent damage	200
Total	8123

Microorganisms and invertebrates

Pesticides easily find their way into soils, where they may be toxic to arthropods, earthworms, fungi, bacteria, and protozoa. Small organisms are vital to ecosystems because they dominate both the structure and function of natural systems.

For example, an estimated 4.5 tons/ha of fungi and bacteria exist in the upper 15 cm of soil. They, with the arthropods, make up 95% of all species and 98% of the biomass (excluding vascular plants). The microorganisms are essential to proper functioning of the ecosystem because they break down organic matter, enabling the vital chemical elements to be recycled (Atlas and Bartha 1987). Equally important is their ability to fix nitrogen, making it available for plants. The role of microorganisms cannot be overemphasized, because in nature, agriculture, and forestry they are essential agents in biogeochemical recycling of the vital elements in all ecosystems (Brock and Madigan 1988).

Although these invertebrates and microorganisms are essential to the vital structure and function of all ecosystems, it is impossible to place a dollar value on the damage caused by pesticides to this large group. To date, no relevant quantitative data has been collected for use in estimating the value of the microorganisms destroyed.

Government funds for pesticide-pollution control

A major environmental cost associated with all pesticide use is the cost of carrying out state and federal regulatory actions, as well as the pesticide monitoring programs needed to control pesticide pollution. Specifically, these funds are spent to reduce the hazards of pesticides and to protect the integrity of the public health and the environment.

At least $1 million is spent each year by the state and federal government to train and register pesticide applicators.[25] Also, more than $40 million is spent each year by EPA for just registering and re-registering pesticides (GAO 1986). We estimate that the federal and state governments together spend approximately $200 million/yr for pesticide pollution control (Table 4).

Although enormous amounts of government money is currently being

[23]R. Beiswenger, 1990, personal communication. University of Wyoming, Laramie.
[24]See footnote 23.

[25]D. Rutz, 1991, personal communication. Department of Entomology, Cornell University, Ithaca, NY.

spent to reduce pesticide pollution, costly damage still results. Also, many serious environmental and social problems remain to be corrected by improved government policies. A recent survey by Sachs et al. (1987) confirmed Sachs' data that confidence in the ability of the US government to regulate pesticides declined from 98% in 1965 to only 46% in 1985. Another survey conducted by the Food and Drug Administration (1989) found that 97% of the public were genuinely concerned that pesticides contaminate their food.

Conclusions

An investment of approximately $4 billion dollars in pesticide control saves approximately $16 billion in US crops, based on direct costs and benefits (Pimentel et al. 1991). However, the indirect environmental and public-health costs of pesticide use need to be balanced against these benefits. Based on the available data, the environmental and social costs of pesticide use total approximately $8 billion each year (Table 4). Users of pesticides in agriculture pay directly for only approximately $3 billion of this cost, which includes problems arising from pesticide resistance and destruction of natural enemies. Society eventually pays this $3 billion plus the remaining $5 billion in environmental and public health costs (Table 4).

Our assessment of the environmental and health problems associated with pesticides is incomplete because data are scarce. What is an acceptable monetary value for a human life lost or for a cancer illness due to pesticides? Equally difficult is placing a monetary value on wild birds and other wildlife, invertebrates, microbes, food, or groundwater.

In addition to the costs that cannot be accurately measured, there are additional costs that have not been included in the $8 billion/yr. A complete accounting of the indirect costs should include accidental poisonings like the aldicarb/watermelon crisis; domestic animal poisonings; unrecorded losses of fish and wildlife and of crops, trees, and other plants; losses resulting from the destruction of soil invertebrates, microflora, and microfauna; true monetary costs of human pesticide poisonings; water and soil pollution; and human health effects such as cancer and sterility. If the full environmental and social costs could be measured as a whole, the total cost would be significantly greater than the estimate of $8 billion/yr. Such a complete long-term cost/benefit analysis of pesticide use would reduce the perceived profitability of pesticides.

Human pesticide poisonings, reduced natural enemy populations, increased pesticide resistance, and honeybee poisonings account for a substantial portion of the calculated environmental and social costs of pesticide use in the United States. Fortunately, some losses of natural enemies and some pesticide resistance problems are being alleviated through carefully planned use of integrated pest management practices. But a great deal remains to be done to reduce these important environmental costs (Pimentel et al. 1991).

The major environmental and public health problems associated with pesticides are in large measure responsible for the loss of public confidence in state and federal regulatory agencies as well as in institutions that conduct agricultural research. Public concern about pesticide pollution confirms a national trend toward environmental values. Media emphasis on the issues and problems caused by pesticides has contributed to a heightened public awareness of ecological concerns. This awareness is encouraging research in environmentally sound agriculture, including non-chemical pest management.

This investigation not only underscores the serious nature of the environmental and socioeconomic costs of pesticides, but it emphasizes the great need for more detailed investigation of the environmental and economic impacts of pesticides. Pesticides are and will continue to be a valuable pest control tool. Meanwhile, with more accurate, realistic cost/benefit analyses, we will be able to work to minimize the risks and to develop and increase the use of nonchemical pest controls to maximize the benefits of pest control strategies for all society.

Acknowledgments

We thank the following people for reading an earlier draft of this manuscript, for their many helpful suggestions, and, in some cases, for providing additional information: A. Blair, National Institutes of Health; J. Blondell, US Environmental Protection Agency; S. A. Briggs, Rachel Carson Council; L. E. Ehler, University of California, Davis; E. L. Flickinger, US Fish and Wildlife Service; T. Frisch, NYCAP; E. L. Gunderson, US Food and Drug Administration; R. G. Hartzler, Iowa State University, Ames; H. Lehman and G. A. Surgeoner, University of Guelph; P. Mineau, Environment Canada; I. N. Oka, Bogor Food and Agriculture Institute, Bogor, Indonesia; C. Osteen, US Department of Agriculture; O. Pettersson, Swedish University of Agricultural Sciences; D. Rosen, Hebrew University of Jerusalem, Jerusalem, Israel; J. Q. Rowley, Oxfam, UK; P. A. Thomson and J. J. Jenkins, Oregon State University; C. Walters, Acres, U.S.A.; G. W. Ware, University of Arizona; and D. H. Beerman, T. Brown, E. L. Madsen, R. Roush, and C. R. Smith, Cornell University.

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Renewable Energy: Economic and Environmental Issues

Solar energy technologies, paired with energy conservation, have the potential to meet a large portion of future US energy needs

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The United States faces serious energy shortages in the near future. High energy consumption and the ever-increasing US population will force residents to confront the critical problem of dwindling domestic fossil energy supplies. With only 4.7% of the world's population, the United States consumes approximately 25% of the total fossil fuel used each year throughout the world. The United States now imports about one half of its oil (25% of total fossil fuel) at an annual cost of approximately $65 billion (USBC 1992a). Current US dependence on foreign oil has important economic costs (Gibbons and Blair 1991) and portends future negative effects on national security and the economy.

Domestic fossil fuel reserves are being rapidly depleted, and it would be a major drain on the economy to import 100% of US oil. Within a decade or two US residents will be forced to turn to renewable energy for some of their energy needs. Proven US oil reserves are projected to be exhausted in 10 to 15 years depending on consumption patterns (DOE 1991a, Matare 1989, Pimentel et al. 1994, Worldwatch Institute 1992), and natural gas reserves are expected to last slightly longer. In contrast, coal reserves have been projected to last approximately 100 years, based on current use and available extraction processes (Matare 1989).

An immediate priority is to speed the transition to renewable, especially solar-based, energy technologies

The US coal supply, however, could be used up in a much shorter period than the projected 100 years, if one takes into account predicted oil and gas depletion and concurrent population growth (DOE 1991a, Matare 1989). The US population is projected to double to more than one-half billion within the next 60 years (USBC 1992b). How rapidly the coal supply is depleted will depend on energy consumption rates. The rapid depletion of US oil and gas reserves is expected to necessitate increased use of coal. By the year 2010, coal may constitute as much as 40% of total energy use (DOE 1991a). Undoubtedly new technologies will be developed that will make it possible to extract more oil and coal. However, this extra extraction can only be achieved at greater energy and economic costs. When the energy input needed to power these methods approaches the amount of energy mined, extraction will no longer be energy cost-effective (Hall et al. 1986).

Fossil fuel combustion, especially that based on oil and coal, is the major contributor to increasing carbon dioxide concentration in the atmosphere, thereby contributing to probable global warming. This climate change is considered one of the most serious environmental threats throughout the world because of its potential impact on food production and processes vital to a productive environment. Therefore, concerns about carbon dioxide emissions may discourage widespread dependence on coal use and encourage the development and use of renewable energy technologies.

Even if the rate of increase of per capita fossil energy consumption is slowed by conservation measures, rapid population growth is expected to speed fossil energy depletion and intensify global warming. Therefore, the projected availability of all fossil energy reserves probably has been overstated. Substantially reducing US use of fossil fuels through the efficient use of energy and the adoption of solar energy technologies extends the life of fossil fuel resources and could provide the time needed to develop and improve renewable energy technologies.

Renewable energy technologies will introduce new conflicts. For example, a basic parameter control-

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Table 1. Current annual fossil (including nuclear) and solar energy use in the United States (DOE 1991a,b, IEA 1991).

Energy type	Quads	Percentage
Fossil energy	78.5	92.3
Solar energy	6.6	7.7
Hydropower	3.0	3.5
Biomass	3.6	4.2
Fuelwood	3.53	4.1
Crop residues	0.07	0.1
Total energy	85.1	100

ling renewable energy supplies is the availability of land. At present more than 99% of the US and world food supply comes from the land (FAO 1991). In addition, the harvest of forest resources is presently insufficient to meet US needs and thus the United States imports some of its forest products (USBC 1992a). With approximately 75% of the total US land area exploited for agriculture and forestry, there is relatively little land available for other uses, such as biomass production and solar technologies. Population growth is expected to further exacerbate the demands for land. Therefore, future land conflicts could be intense.

In this article, we analyze the potential of various renewable or solar energy technologies to supply the United States with its future energy needs. Diverse renewable technologies are assessed in terms of their land requirements, environmental benefits and risks, economic costs, and a comparison of their advantages. In addition, we make a projection of the amount of energy that could be supplied by solar energy subject to the constraints of maintaining the food and forest production required by society. Although renewable energy technologies often cause fewer environmental problems than fossil energy systems, they require large amounts of land and therefore compete with agriculture, forestry, and other essential land-use systems in the United States.

Assessment of renewable energy technologies

Coal, oil, gas, nuclear, and other mined fuels currently provide most of US energy needs. Renewable energy technologies provide only 8% (Table 1).

The use of solar energy is, however, expected to grow. Renewable energy technologies that have the potential to provide future energy supplies include: biomass systems, hydroelectric systems, hydrogen fuel, wind power, photovoltaics, solar thermal systems, and passive and active heating and cooling systems.

Biomass energy systems

At present, forest biomass energy, harvested from natural forests, provides an estimated 3.6 quads (1.1 x 10^{18} Joules) or 4.2% of the US energy supply (Table 1). Worldwide, and especially in developing countries, biomass energy is more widely used than in the United States. Only forest biomass will be included in this US assessment, because forest is the most abundant biomass resource and the most concentrated form of biomass. However, some biomass proponents are suggesting the use of grasses, which on productive soils can yield an average of 5 $t \bullet ha^{-1} \bullet yr^{-1}$ (Hall et al. 1993, USDA 1992).

Although in the future most biomass probably will be used for space and water heating, we have analyzed its conversion into electricity in order to clarify the comparison with other renewable technologies. An average of 3 tons of (dry) woody biomass can be sustainably harvested per hectare per year with small amounts of nutrient fertilizer inputs (Birdsey 1992). This amount of woody biomass has a gross energy yield of 13.5 million kcal (thermal). The net yield is, however, lower because approximately 33 liters of diesel fuel oil per hectare is expended for cutting and collecting wood and for transportation, assuming an 80-kilometer roundtrip between the forest and the plant. The economic benefits of biomass are maximized when biomass can be used close to where it is harvested.

A city of 100,000 people using the biomass from a sustainable forest (3 tons/ha) for fuel would require approximately 220,000 ha of forest area, based on an electrical demand of 1 billion kWh (860 x 10^9 kcal = 1 kWh) per year (Table 2). Nearly 70% of the heat energy produced from burning biomass is lost in the conversion into electricity, similar to losses experienced in coal-fired plants. The area required is about the same as that currently used by 100,000 people for food production, housing, industry, and roadways (USDA 1992).

The energy input/output ratio of this system is calculated to be 1:3 (Table 2). The cost of producing a kilowatt of electricity from woody biomass ranges from 7¢ to 10¢ (Table 2), which is competitive for electricity production that presently has a cost ranging from 3¢ to 13¢ (Table 2; USBC 1992a). Approximately 3 kcal of thermal energy is required to produce 1 kcal of electricity. Biomass could supply the nation with 5 quads of its total gross energy supply by the year 2050 with the use of at least 75 million ha (an area larger than Texas, or approximately 8% of the 917 million ha in the United States) (Table 3).

However, several factors limit reliance on woody biomass. Certainly, culturing fast-growing trees in a plantation system located on prime land might increase yields of woody biomass. However, this practice is unrealistic because prime land is essential for food production. Furthermore, such intensely managed systems require additional fossil fuel inputs for heavy machinery, fertilizers, and pesticides, thereby diminishing the net energy available. In addition, Hall et al. (1986) point out that energy is not the highest priority use of trees.

If natural forests are managed for maximal biomass energy production, loss of biodiversity can be expected. Also, the conversion of natural forests into plantations increases soil erosion and water runoff. Continuous soil erosion and degradation would ultimately reduce the overall productivity of the land. Despite serious limitations of plantations, biomass production could be increased using agroforestry technologies designed to protect soil quality and conserve biodiversity. In these systems, the energy and economic costs would be significant and therefore might limit the use of this strategy.

The burning of biomass is environmentally more polluting than gas but less polluting than coal. Bio-

Table 2. Land resource requirements and total energy inputs for construction of solar and other energy facilities that produce 1 billion kWh/yr of electricity (averaged over 30-year life of unit). Energy return on investment is listed for each technology.

Electrical energy technology	Land required (ha)	Energy required (kWh x 10^9)	Energy return on investment	Cost per kWh (¢)
Hydroelectric	75,000*	0.021†	48:1	2‡
Biomass	220,000	0.3	3:1	7-10‡
Central receivers	1,100§	0.1§	10:1	10‖
Solar ponds	5,200¶	0.248	4:1	14
Wind power	11,666#	0.205	5:1	6
Photovoltaics	2,700**	0.108	9:1	30
Coal	363††	0.12	8:1	3
Nuclear	48††	0.20	5:1	5

* Based on a random sample of 50 hydropower reservoirs in the United States, ranging in area from 482 ha to 763,000 ha (FERC 1984, ICLD 1988).
† Pimentel et al. 1994.
‡ Based on personal communication (1992) with John Irving, station superindendent, City of Burlington Electric Light Department (Vermont).
§ For three 100 MW plants (Vant-Hull 1992).
‖ Vant-Hull 1992, Williams et al. 1990.
¶ Based on 4000 ha solar ponds plus an additional 1200 ha for evaporation ponds.
\# AWEA 1991.
** Calculated from Hesperia operational statistics in 1989 (EPRI 1991).
†† Smil 1984.

mass combustion releases more than 100 different chemical pollutants into the atmosphere (Alfheim and Ramdahl 1986). Wood smoke is reported to contain pollutants known to cause bronchitis, emphysema, and other illnesses. These pollutants include up to 14 carcinogens, 4 cocarcinogens, 6 toxins that damage cilia, and additional mucus-coagulating agents (Alfheim and Ramdahl 1986, DOE 1980). Of special concern are the relatively high concentrations of potentially carcinogenic polycyclic aromatic hydrocarbons (PAHs, organic compounds such as benzo(a)pyrene) and particulates found in biomass smoke (DOE 1980). Sulfur and nitrogen oxides, carbon monoxide, and aldehydes also are released in small though significant quantities and contribute to reduced air quality (DOE 1980). In electric-generating plants, however, as much as 70% of these air pollutants can be removed by installing the appropriate air-pollution control devices in the combustion system.

Because of pollutants, several communities (including Aspen, Colorado) have banned the burning of wood for heating homes. When biomass is burned continuously in the home for heating, its pollutants can be a threat to human health (Lipfert et al. 1988, Smith 1987b).

When biomass in the form of harvested crop residues is used for fuel, the soil is exposed to intense erosion by wind and water (Pimentel et al. 1984). In addition to the serious degradation of valuable agricultural land, the practice of burning crop residues as a fuel removes essential nutrients from the land and requires the application of costly fossil-based fertilizers if yields are to be maintained. However, the soil organic matter, soil biota, and water-holding capacity of the soil cannot be replaced by applying fertilizers. Therefore, we conclude that crop residues should not be removed from the land for a fuel source (Pimentel 1992).

Biomass will continue to be a valuable renewable energy resource in the future, but its expansion will be greatly limited. Its use conflicts with the needs of agricultural and forestry production and contributes to major environmental problems.

Liquid fuels

Liquid fuels are indispensable to the US economy (DOE 1991a). Petroleum, essential for the transportation sector as well as the chemical industry, makes up approximately 42% of total US energy consumption. At present, the United States imports about one half of its petroleum and is projected to import nearly 100% within 10 to 15 years (DOE 1991a). Barring radically improved electric battery technologies, a shift from petroleum to alternative liquid and gaseous fuels will have to be made. The analysis in this section is focused on the potential of three liquid fuels: ethanol, methanol, and hydrogen.

Ethanol. A wide variety of starch and sugar crops, food processing wastes, and woody materials (Lynd et al. 1991) have been evaluated as raw materials for ethanol production. In the United States, corn appears to be the most feasible biomass feedstock in terms of availability and technology (Pimentel 1991).

The total fossil energy expended to produce 1 liter of ethanol from corn is 10,200 kcal, but note that 1 liter of ethanol has an energy value of only 5130 kcal. Thus, there is an energy imbalance causing a net energy loss. Approximately 53% of the total cost (55¢ per liter) of producing ethanol in a large, modern plant is for the corn raw material (Pimentel 1991). The total energy inputs for producing ethanol using corn can be partially offset when the dried distillers grain produced is fed to livestock. Although the feed value of the dried distillers grain reduces the total energy inputs by 8% to 24%, the energy budget remains negative.

The major energy input in ethanol production, approximately 40% overall, is fuel needed to run the distillation process (Pimentel 1991). This fossil energy input contributes to a negative energy balance and atmospheric pollution. In the production process, special membranes can separate the ethanol from the so-called beer produced by fermentation. The most promising systems rely on distillation to bring the ethanol concentration up to 90%, and selective-membrane processes are used to further raise the ethanol concentration to 99.5% (Maeda and Kai 1991). The energy input for this upgrading is approximately 1280 kcal/liter. In laboratory tests, the total input for producing a liter of ethanol can potentially be reduced from 10,200 to 6200 kcal by using

membranes, but even then the energy balance remains negative.

Any benefits from ethanol production, including the corn by-products, are negated by the environmental pollution costs incurred from ethanol production (Pimentel 1991). Intensive corn production in the United States causes serious soil erosion and also requires the further draw-down of groundwater resources. Another environmental problem is caused by the large quantity of stillage or effluent produced. During the fermentation process approximately 13 liters of sewage effluent is produced and placed in the sewage system for each liter of ethanol produced.

Although ethanol has been advertised as reducing air pollution when mixed with gasoline or burned as the only fuel, there is no reduction when the entire production system is considered. Ethanol does release less carbon monoxide and sulfur oxides than gasoline and diesel fuels. However, nitrogen oxides, formaldehydes, other aldehydes, and alcohol—all serious air pollutants—are associated with the burning of ethanol as fuel mixture with or without gasoline (Sillman and Samson 1990). Also, the production and use of ethanol fuel contribute to the increase in atmospheric carbon dioxide and to global warming, because twice as much fossil energy is burned in ethanol production than is produced as ethanol.

Ethanol produced from corn clearly is not a renewable energy source. Its production adds to the depletion of agricultural resources and raises ethical questions at a time when food supplies must increase to meet the basic needs of the rapidly growing world population.

Methanol. Methanol is another potential fuel for internal combustion engines (Kohl 1990). Various raw materials can be used for methanol production, including natural gas, coal, wood, and municipal solid wastes. At present, the primary source of methanol is natural gas. The major limitation in using biomass for methanol production is the enormous quantities needed for a plant with suitable economies of scale. A suitably large methanol plant would require at least 1250 tons of dry biomass per day for processing (ACTI 1983). More than 150,000 ha of forest would be needed to supply one plant. Biomass generally is not available in such enormous quantities from extensive forests and at acceptable prices (ACTI 1983).

If methanol from biomass (33 quads) were used as a substitute for oil in the United States, from 250 to 430 million ha of land would be needed to supply the raw material. This land area is greater than the 162 million ha of US cropland now in production (USDA 1992). Although methanol production from biomass may be impractical because of the enormous size of the conversion plants (Kohl 1990), it is significantly more efficient than the ethanol production system based on both energy output and economics (Kohl 1990).

Compared to gasoline and diesel fuel, both methanol and ethanol reduce the amount of carbon monoxide and sulfur oxide pollutants produced, however both contribute other major air pollutants such as aldehydes and alcohol. Air pollutants from these fuels worsen the tropospheric ozone problem because of the emissions of nitrogen oxides from the richer mixtures used in the combustion engines (Sillman and Samson 1990).

Table 3. Current and projected US gross annual energy supply from various solar energy technologies based on thermal equivalents and required land area.

Energy technology	Current (1992) Quads	Current (1992) Million ha	Projected (2050) Quads	Projected (2050) Million ha
Biomass	3.6	100* (50†)	5	163* (75†)
Hydroelectric	2.9‡	63§	4	87
Solar thermal	< 0.001	< 0.001	6	5
Photovoltaics	< 0.001	< 0.001	8	6
Wind power	0.01‖	0.5	8¶	9#
Passive solar	0.3**	0	6	1
Total solar energy	7	164 (114†)	37	271 (173†)

* This is the equivalent land area required to produce 3.5 t/ha plus the energy required for harvesting and transport. However, we estimate that 60% of this biomass is coming from forest wastes and residues. Thus, only approximately 40% of the total land is assumed to be managed for forest biomass energy.
† See above footnote*.
‡ USBC 1992a.
§ Total area based on an average of 75,000 ha per reservoir area per 1 billion kWh/yr produced (Table 2).
‖ AWEA 1992.
¶ Ranges from a projected 3–4 quads (DOE 1992) to 11 quads (DOE 1990).
\# If the potential dual use of the land for agricultural purposes is considered, then it might be possible to reduce the land area by approximately 95%.
** Flavin 1985.

Hydrogen. Gaseous hydrogen, produced by the electrolysis of water, is another alternative to petroleum fuels. Using solar electric technologies for its production, hydrogen has the potential to serve as a renewable gaseous and liquid fuel for transportation vehicles. In addition hydrogen can be used as an energy storage system for electrical solar energy technologies, like photovoltaics (Winter and Nitsch 1988).

The material inputs for a hydrogen production facility are primarily those needed to build a solar electric production facility. The energy required to produce 1 billion kWh of hydrogen is 1.3 billion kWh of electricity (Voigt 1984). If current photovoltaics (Table 2) require 2700 ha/1 billion kWh, then a total area of 3510 ha would be needed to supply the equivalent of 1 billion kWh of hydrogen fuel. Based on US per capita liquid fuel needs, a facility covering approximately 0.15 ha (16,300 ft^2) would be needed to produce a year's requirement of liquid hydrogen. In such a facility, the water requirement for electrolytic production of 1 billion kWh/yr equivalent of hydrogen is approximately 300 million liters/yr (Voigt 1984).

To consider hydrogen as a substitute for gasoline: 9.5 kg of hydrogen produces energy equivalent to that produced by 25 kg of gasoline. Stor-

ing 25 kg of gasoline requires a tank with a mass of 17 kg, whereas the storage of 9.5 kg of hydrogen requires 55 kg (Peschka 1987). Part of the reason for this difference is that the volume of hydrogen fuel is about four times greater than that for the same energy content of gasoline. Although the hydrogen storage vessel is large, hydrogen burns 1.33 times more efficiently than gasoline in automobiles (Bockris and Wass 1988). In tests, a BMW 745i liquid-hydrogen test vehicle with a tank weight of 75 kg, and the energy equivalent of 40 liters (320,000 kcal) of gasoline, had a cruising range in traffic of 400 km or a fuel efficiency of 10 km per liter (24 mpg) (Winter 1986).

At present, commercial hydrogen is more expensive than gasoline. For example, assuming 5¢ per kWh of electricity from a conventional power plant, hydrogen would cost 9¢ per kWh (Bockris and Wass 1988). This cost is the equivalent of 67¢/liter of gasoline. Gasoline sells at the pump in the United States for approximately 30¢/liter. However, estimates are that the real cost of burning a liter of gasoline ranges from $1.06 to $1.32, when production, pollution, and other external costs are included (Worldwatch Institute 1989). Therefore, based on these calculations hydrogen fuel may eventually be competitive.

Some of the oxygen gas produced during the electrolysis of water can be used to offset the cost of hydrogen. Also the oxygen can be combined with hydrogen in a fuel cell, like those used in the manned space flights. Hydrogen fuel cells used in rural and suburban areas as electricity sources could help decentralize the power grid, allowing central power facilities to decrease output, save transmission costs, and make mass-produced, economical energy available to industry.

Compared with ethanol, less land (0.15 ha versus 7 ha for ethanol) is required for hydrogen production that uses photovoltaics to produce the needed electricity. The environmental impacts of hydrogen are minimal. The negative impacts that occur during production are all associated with the solar electric technology used in production. Water for the production of hydrogen may be a problem in the arid regions of the United States, but the amount required is relatively small compared with the demand for irrigation water in agriculture. Although hydrogen fuel produces emissions of nitrogen oxides and hydrogen peroxide pollutants, the amounts are about one-third lower than those produced from gasoline engines (Veziroglu and Barbir 1992). Based on this comparative analysis, hydrogen fuel may be a cost-effective alternative to gasoline, especially if the environmental and subsidy costs of gasoline are taken into account.

Hydroelectric systems

For centuries, water has been used to provide power for various systems. Today hydropower is widely used to produce electrical energy. In 1988 approximately 870 billion kWh (3 quads or 9.5%) of the United States' electrical energy was produced by hydroelectric plants (FERC 1988, USBC 1992a). Further development and/or rehabilitation of existing dams could produce an additional 48 billion kWh per year. However, most of the best candidate sites already have been fully developed, although some specialists project increasing US hydropower by as much as 100 billion kWh if additional sites are developed (USBC 1992a).

Hydroelectric plants require land for their water-storage reservoirs. An analysis of 50 hydroelectric sites in the United States indicated that an average of 75,000 ha of reservoir area are required per 1 billion kWh/yr produced (Table 2). However, the size of reservoir per unit of electricity produced varies widely, ranging from 482 ha to 763,000 ha per 1 billion kWh/yr depending upon the hydro head, terrain, and additional uses made of the reservoir (Table 2). The latter include flood control, storage of water for public and irrigation supplies, and/or recreation (FERC 1984). For the United States the energy input/output ratio was calculated to be 1:48 (Table 2); for Europe an estimate of 1:15 has been reported (Winter et al. 1992).

Based on regional estimates of land use and average annual energy generation, approximately 63 million hectares of the total of 917 million ha of land area in the United States are currently covered with reservoirs. To develop the remaining best candidate sites, assuming land requirements similar to those in past developments, an additional 24 million hectares of land would be needed for water storage (Table 3).

Reservoirs constructed for hydroelectric plants have the potential to cause major environmental problems. First, the impounded water frequently covers agriculturally productive, alluvial bottomland. This water cover represents a major loss of productive agricultural land. Dams may fail, resulting in loss of life and destruction of property. Further, dams alter the existing plant and animal species in the ecosystem (Flavin 1985). For example, cold-water fishes may be replaced by warmwater fishes, frequently blocking fish migration (Hall et al. 1986). However, flow schedules can be altered to ameliorate many of these impacts. Within the reservoirs, fluctuations of water levels alter shorelines and cause downstream erosion and changes in physiochemical factors, as well as the changes in aquatic communities. Beyond the reservoirs, discharge patterns may adversely reduce downstream water quality and biota, displace people, and increase water evaporation losses (Barber 1993). Because of widespread public environmental concerns, there appears to be little potential for greatly expanding either large or small hydroelectric power plants in the future (Table 3).

Wind power

For many centuries, wind power like water power has provided energy to pump water and run mills and other machines. In rural America windmills have been used to generate electricity since the early 1900s.

Modern wind turbine technology has made significant advances over the last 10 years. Today, small wind machines with 5 to 40 kW capacity can supply the normal electrical needs of homes and small industries (Twidell 1987). Medium-size turbines rated 100 kW to 500 kW pro-

duce most of the commercially generated electricity. At present, the larger, heavier blades required by large turbines upset the desirable ratio between size and weight and create efficiency problems. However, the effectiveness and efficiency of the large wind machines are expected to be improved through additional research and development of lighter weight but stronger components (Clarke 1991). Assuming a 35% operation capacity at a favorable site, the energy input/output ratio of the system is 1:5 for the material used in the construction of medium-size wind machines (Table 2).

The availability of sites with sufficient wind (at least 20 km/h) limits the widespread development of wind farms. Currently, 70% of the total wind energy (0.01 quad) produced in the United States is generated in California (Table 3; AWEA 1992). However, an estimated 13% of the contiguous US land area has wind speeds of 22 km/h or higher; this area then would be sufficient to generate approximately 20% of US electricity using current technology (DOE 1992). Promising areas for wind development include the Great Plains and coastal regions.

Another limitation of this energy resource is the number of wind machines that a site can accommodate. For example, at Altamont Pass, California, an average of one turbine per 1.8 ha allows sufficient spacing to produce maximum power (Smith and Ilyin 1991). Based on this figure approximately 11,700 ha of land are needed to supply 1 billion kWh/yr (Table 2). However, because the turbines themselves only occupy approximately 2% of the area or 230 ha, dual land use is possible. For example, current agricultural land developed for wind power continues to be used in cattle, vegetable, and nursery stock production.

An investigation of the environmental impacts of wind energy production reveals a few hazards. For example, locating the wind turbines in or near the flyways of migrating birds and wildlife refuges may result in birds flying into the supporting structures and rotating blades (Clarke 1991, Kellett 1990). Clarke suggests that wind farms be located at least 300 meters from nature reserves to reduce this risk to birds.

Insects striking turbine blades will probably have only a minor impact on insect populations, except for some endangered species. However, significant insect accumulation on the blades may reduce turbine efficiency (Smith 1987a).

Wind turbines create interference with electromagnetic transmission, and blade noise may be heard up to 1 km away (Kellet 1990). Fortunately, noise and interference with radio and television signals can be eliminated by appropriate blade materials and careful placement of turbines. In addition, blade noise is offset by locating a buffer zone between the turbines and human settlements. New technologies and designs may minimize turbine generator noise.

Under certain circumstances shadow flicker has caused irritation, disorientation, and seizures in humans (Steele 1991). However, as with other environmental impacts, mitigation is usually possible through careful site selection away from homes and offices. This problem slightly limits the land area suitable for wind farms.

Although only a few wind farms supply power to utilities in the United States, future widespread development may be constrained because local people feel that wind farms diminish the aesthetics of the area (Smith 1987a). Some communities have even passed legislation to prevent wind turbines from being installed in residential areas (Village of Cayuga Heights, New York, Ordinance 1989). Likewise areas used for recreational purposes, such as parks, limit the land available for wind power development.

Photovoltaics

Photovoltaic cells are likely to provide the nation with a significant portion of its future electrical energy (DeMeo et al. 1991). Photovoltaic cells produce electricity when sunlight excites electrons in the cells. Because the size of the units is flexible and adaptable, photovoltaic cells are ideal for use in homes, industries, and utilities.

Before widespread use, however, improvements are needed in the photovoltaic cells to make them economically competitive. Test photovoltaic cells that consist of silicon solar cells are currently up to 21% efficient in converting sunlight into electricity (Moore 1992). The durability of photovoltaic cells, which is now approximately 20 years, needs to be lengthened and current production costs reduced about fivefold to make them economically feasible. With a major research investment, all of these goals appear possible to achieve (DeMeo et al. 1991).

Currently, production of electricity from photovoltaic cells costs approximately 30¢/kWh, but the price is projected to fall to approximately 10¢/kWh by the end of the decade and perhaps reach as low as 4¢ by the year 2030, provided the needed improvements are made (Flavin and Lenssen 1991). In order to make photovoltaic cells truly competitive, the target cost for modules would have to be approximately 8¢/kWh (DeMeo et al. 1991).

Using photovoltaic modules with an assumed 7.3% efficiency (the current level of commercial units), 1 billion kWh/yr of electricity could be produced on approximately 2700 ha of land (Table 2), or approximately 0.027 ha per person, based on the present average per capita use of electricity. Thus, total US electrical needs theoretically could be met with photovoltaic cells on 5.4 million ha (0.6% of US land). If 21% efficient cells were used, the total area needed would be greatly reduced. Photovoltaic plants with this level of efficiency are being developed (DeMeo et al. 1991).

The energy input for the structural materials of a photovoltaic system delivering 1 billion kWh is calculated to be approximately 300 kWh/m^2. The energy input/output ratio for production is about 1:9 assuming a life of 20 years (Table 2).

Locating the photovoltaic cells on the roofs of homes, industries, and other buildings would reduce the need for additional land by approximately 5% (USBC 1992a), as well as reduce the costs of energy transmission. However, photovoltaic systems require back-up with conventional electrical systems, be-

cause they function only during daylight hours.

Photovoltaic technology offers several environmental advantages in producing electricity compared with fossil fuel technologies. For example, using present photovoltaic technology, carbon dioxide emissions and other pollutants are negligible.

The major environmental problem associated with photovoltaic systems is the use of toxic chemicals such as cadmium sulfide and gallium arsenide, in their manufacture (Holdren et al. 1980). Because these chemicals are highly toxic and persist in the environment for centuries, disposal of inoperative cells could become a major environmental problem. However, the most promising cells in terms of low cost, mass production, and relatively high efficiency are those being manufactured using silicon. This material makes the cells less expensive and environmentally safer than the heavy metal cells.

Solar thermal conversion systems

Solar thermal energy systems collect the sun's radiant energy and convert it into heat. This heat can be used for household and industrial purposes and also to drive a turbine and produce electricity. System complexity ranges from solar ponds to the electric-generating central receivers. We have chosen to analyze electricity in order to facilitate comparison to the other solar energy technologies.

Solar ponds. Solar ponds are used to capture solar radiation and store it at temperatures of nearly 100°C. Natural or man-made ponds can be made into solar ponds by creating a salt-concentration gradient made up of layers of increasing concentrations of salt. These layers prevent natural convection from occurring in the pond and enable heat collected from solar radiation to be trapped in the bottom brine.

The hot brine from the bottom of the pond is piped out for generating electricity. The steam from the hot brine turns freon into a pressurized vapor, which drives a Rankine engine. This engine was designed specifically for converting low-grade heat into electricity. At present, solar ponds are being used in Israel to generate electricity (Tabor and Doran 1990).

For successful operation, the salt-concentration gradient and the water levels must be maintained. For example, 4000 ha of solar ponds lose approximately 3 billion liters of water per year under the arid conditions of the southwestern United States (Tabor and Doran 1990). In addition, to counteract the water loss and the upward diffusion process of salt in the ponds, the dilute salt water at the surface of the ponds has to be replaced with fresh water. Likewise salt has to be added periodically to the heat-storage zone. Evaporation ponds concentrate the brine, which can then be used for salt replacement in the solar ponds.

Approximately 4000 ha of solar ponds (40 ponds of 100 ha) and a set of evaporation ponds that cover a combined 1200 ha are needed for the production of 1 billion kWh of electricity needed by 100,000 people in one year (Table 2). Therefore, a family of three would require approximately 0.2 ha (22,000 sq ft) of solar ponds for its electricity needs. Although the required land area is relatively large, solar ponds have the capacity to store heat energy for days, thus eliminating the need for back-up energy sources from conventional fossil plants. The efficiency of solar ponds in converting solar radiation into heat is estimated to be approximately 1:5. Assuming a 30-year life for a solar pond, the energy input/output ratio is calculated to be 1:4 (Table 2). A 100-hectare (1 km^2) solar pond is calculated to produce electricity at a rate of approximately 14¢ per kWh. According to Folchitto (1991), this cost could be reduced in the future.

In several locations in the United States solar ponds are now being used successfully to generate heat directly. The heat energy from the pond can be used to produce processed steam for heating at a cost of only 2¢ to 3.5¢ per kWh (Gommend and Grossman 1988). Solar ponds are most effectively employed in the Southwest and Mid-west.

Some hazards are associated with solar ponds, but most can be prevented with careful management. For instance, it is essential to use plastic liners to make the ponds leakproof and thereby prevent contamination of the adjacent soil and groundwater with salt. Burrowing animals must be kept away from the ponds by buried screening (Dickson and Yates 1983). In addition, the ponds should be fenced to prevent people and other animals from coming in contact with them. Because some toxic chemicals are used to prevent algae growth on water surface and freon is used in the Rankine engine, methods will have to be devised for safely handling these chemicals (Dickson and Yates 1983).

Solar receiver systems. Other solar thermal technologies that concentrate solar radiation for large scale energy production include distributed and central receivers. Distributed receiver technologies use rows of parabolic troughs to focus sunlight on a central-pipe receiver that runs above the troughs. Pressurized water and other fluids are heated in the pipe and are used to generate steam to drive a turbogenerator for electricity production or provide industry with heat energy.

Central receiver plants use computer-controlled, sun-tracking mirrors, or heliostats, to collect and concentrate the sunlight and redirect it toward a receiver located atop a centrally placed tower. In the receiver, the solar energy is captured as heat energy by circulating fluids, such as water or molten salts, that are heated under pressure. These fluids either directly or indirectly generate steam, which is then driven through a conventional turbogenerator to yield electricity. The receiver system may also be designed to generate heat for industry.

Distributed receivers have entered the commercial market before central receivers, because central receivers are more expensive to operate. But, compared to distributed receivers, central receivers have the potential for greater efficiency in electricity production because they are able to achieve higher energy concentrations and higher turbine inlet temperatures (Winter 1991). Central receivers are used in this analysis.

The land requirements for the central receiver technology are approximately 1100 ha to produce 1 billion kWh/yr (Table 2), assuming peak efficiency, and favorable sunlight conditions like those in the western United States. Proposed systems offer four to six hours of heat storage and may be constructed to include a back-up alternate energy source. The energy input/output ratio is calculated to be 1:10 (Table 2). Solar thermal receivers are estimated to produce electricity at approximately 10¢ per kWh, but this cost is expected to be reduced somewhat in the future, making the technology more competitive (Vant-Hull 1992). New technical advances aimed at reducing costs and improving efficiency include designing stretched membrane heliostats, volumetric-air ceramic receivers, and improved overall system designs (Beninga et al. 1991).

Central receiver systems are being tested in Italy, France, Spain, Japan, and the United States (at the 10-megawatt Solar One pilot plant near Barstow, California; Skinrood and Skvarna 1986). Also, Luz's Solar Electric Generating System plants at Barstow use distributed receivers to generate almost 300 MW of commercial electricity (Jensen et al. 1989).

The potential environmental impacts of solar thermal receivers include: the accidental or emergency release of toxic chemicals used in the heat transfer system (Baechler and Lee 1991); bird collisions with a heliostat and incineration of both birds and insects if they fly into the high temperature portion of the beams; and—if one of the heliostats did not track properly but focused its high temperature beam on humans, other animals, or flammable materials—burns, retinal damage, and fires (Mihlmester et al. 1980). Flashes of light coming from the heliostats may pose hazards to air and ground traffic (Mihlmester et al. 1980).

Other potential environmental impacts include microclimate alteration, for example reduced temperature and changes in wind speed and evapotranspiration beneath the heliostats or collecting troughs. This alteration may cause shifts in various plant and animal populations. The albedo in solar-collecting fields may be increased from 30% to 56% in desert regions (Mihlmester et al. 1980). An area of 1100 ha is affected by a plant producing 1 billion kWh.

The environmental benefits of receiver systems are significant when compared to fossil fuel electrical generation. Receiver systems cause no problems of acid rain, air pollution, or global warming (Kennedy et al. 1991).

Passive heating and cooling of buildings

Approximately 23% (18.4 quads) of the fossil energy consumed yearly in the United States is used for space heating and cooling of buildings and for heating hot water (DOE 1991a). At present only 0.3 quads of energy are being saved by technologies that employ passive and active solar heating and cooling of buildings (Table 2). Tremendous potential exists for substantial energy savings by increased energy efficiency and by using solar technologies for buildings.

Both new and established homes can be fitted with solar heating and cooling systems. Installing passive solar systems into the design of a new home is generally cheaper than retrofitting an existing home. Including passive solar systems during new home construction usually adds less than 10% to construction costs (Howard and Szoke 1992); a 3–5% added first cost is typical.[1] Based on the cost of construction and the amount of energy saved measured in terms of reduced heating costs, we estimate the cost of passive solar systems to be approximately 3¢ per kWh saved.

Improvements in passive solar technology are making it more effective and less expensive than in the past. In the area of window designs, for example, current research is focused on the development of superwindows with high-insulating values and smart or electrochromic windows that can respond to electrical current, temperature, or incident sunlight to control the admission of light energy (Warner 1991). Use of transparent insulation materials makes window designs that transmit from 50% to 70% of incident solar energy while at the same time providing insulating values typical of 25 cm of fiber glass insulation (Chahroudi 1992). Such materials have a wide range of solar technology applications beyond windows, including house heating with transparent, insulated collector-storage walls and integrated storage collectors for domestic hot water (Wittwer et al. 1991).

Active solar heating technologies are not likely to play a major role in the heating of buildings. The cost of energy saved is relatively high compared with passive systems and conservation measures.[2]

Solar water heating is also cost-effective. Approximately 3% of all the energy used in the United States is for heating water in homes (DOE 1991a). In addition, many different types of passive and active water heating solar systems are available and are in use throughout the United States. These systems are becoming increasingly affordable and reliable (Wittwer et al. 1991). The cost of purchasing and installing an active solar water heater ranges from $2500 to $6000 in the northern regions and $2000 to $4000 in the southern regions of the nation (DOE/CE 1988).

Although none of the passive heating and cooling technologies require land, they can cause environmental problems. For example, some indirect land-use problems may occur, such as the removal of trees, shading, and rights to the sun (Schurr et al. 1979). Glare from collectors and glazing could create hazards to automobile drivers, pedestrians, bicyclists, and airline pilots. Also, when houses are designed to be extremely energy efficient and airtight, indoor air quality becomes a concern because air pollutants may accumulate inside. However, installation of well-designed ventilation systems promotes a healthful exchange of air while reducing heat loss during the winter and heat gain during the summer. If radon is a pollutant

[1]B. D. Howard, 1992, personal communication. The Alliance to Save Energy, Washington, DC.

[2]See footnote 1.

present at unsafe levels in the home, various technologies can mitigate the problem (ASTM 1992).

Comparing solar power to coal and nuclear power

Coal and nuclear power production are included in this analysis to compare conventional sources of electricity generation to various future solar energy technologies. Coal, oil, gas, nuclear, and other mined fuels are used to meet 92% of US energy needs (Table 1). Coal and nuclear plants combined produce three quarters of US electricity (USBC 1992a).

Energy efficiencies for both coal and nuclear fuels are low due to the thermal law constraint of electric generator designs: coal is approximately 35% efficient and nuclear fuels approximately 33% (West and Kreith 1988). Both coal and nuclear power plants in the future may require additional structural materials to meet clean air and safety standards. However, the energetic requirements of such modifications are estimated to be small compared with the energy lost due to conversion inefficiencies.

The costs of producing electricity using coal and nuclear energy are 3¢ and 5¢ per kWh, respectively (EIA 1990). However, the costs of this kind of energy generation are artificially low because they do not include such external costs as damages from acid rain produced from coal and decommissioning costs for the closing of nuclear plants. The Clean Air Act and its amendments may raise coal generation costs, while the new reactor designs, standardization, and streamlined regulations may reduce nuclear generation costs. Government subsidies for nuclear and coal plants also skew the comparison with solar energy technologies (Wolfson 1991).

Clouding the economic costs of fossil energy use are the direct and indirect US subsidies that hide the true cost of energy and keep the costs low, thereby encouraging energy consumption. The energy industry receives a direct subsidy of $424 per household per year (based on an estimated maximum of $36 billion for total federal energy subsidies [ASE 1993]). In addition, the mined-energy industry, like the gasoline industry, does not pay for the environmental and public health costs of fossil energy production and consumption.

The land requirements for fossil fuel and nuclear-based plants are lower than those for solar energy technologies (Table 2). The land area required for electrical production of 1 billion kWh/year is estimated at 363 ha for coal and 48 ha for nuclear fuels. These figures include the area for the plants and both surface and underground mining operations and waste disposal. The land requirements for coal technology are low because it uses concentrated fuel sources rather than diffuse solar energy. However, as the quality of fuel ore declines, land requirements for mining will increase. In contrast, efficient reprocessing and the use of nuclear breeder reactors may decrease the land area necessary for nuclear power.

Many environmental problems are associated with both coal and nuclear power generation (Pimentel et al. 1994). For coal, the problems include the substantial damage to land by mining, air pollution, acid rain, global warming, as well as the safe disposal of large quantities of ash (Wolfson 1991). For nuclear power, the environmental hazards consist mainly of radioactive waste that may last for thousands of years, accidents, and the decommissioning of old nuclear plants (Wolfson 1991).

Fossil-fuel electric utilities account for two-thirds of the sulfur dioxide, one-third of the nitrogen dioxide, and one-third of the carbon dioxide emissions in the United States (Kennedy et al. 1991). Removal of carbon dioxide from coal plant emissions could raise costs to 12¢/kWh; a disposal tax on carbon could raise coal electricity costs to 18¢/kWh (Williams et al. 1990).

The occupational and public health risks of both coal and nuclear plants are fairly high, due mainly to the hazards of mining, ore transportation, and subsequent air pollution during the production of electricity. However, there are 22 times as many deaths per unit of energy related to coal than of nuclear energy production because 90,000 times greater volume of coal than nuclear ore is needed to generate an equivalent amount of electricity.[3]

Also, and as important, coal produces more diffuse pollutants than nuclear fuels during normal operation of the generating plant. Coal-fired plants produce air pollutants—including sulfur oxides, nitrogen oxides, carbon dioxide, and particulates—that adversely affect air quality and contribute to acid rain. Technologies do exist for removing most of the air pollutants, but their use increases the cost of a new plant by 20–25% (IEA 1987). By comparison, nuclear power produces many fewer pollutants than do coal plants (Tester et al. 1991).

Transition to solar energy and other alternatives

The first priority of a sustainable US energy program should be for individuals, communities, and industries to conserve fossil energy resources. Other developed countries have proven that high productivity and a high standard of living can be achieved with considerably less energy expenditure compared to that of the United States. Improved energy efficiency in the United States, other developed nations, and even in developing nations would help both extend the world's fossil energy resources and improve the environment (Pimentel et al. 1994).

The supply and demand for fossil and solar energy; the requirements of land for food, fiber, and lumber; and the rapidly growing human population will influence future US options. The growth rate of the US population has been increasing and is now at 1.1% per year (USBC 1992b); at this rate, the present population of 260 million will increase to more than a half billion in just 60 years. The presence of more people will require more land for homes, businesses, and roads. Population density directly influences food production, forest product needs, and energy requirements. Considerably more agricultural and forest land will be needed to provide vital food and forest products, and

[3]D. Hammer, 1993, personal communication. Cornell University, Ithaca, NY.

the drain on all energy resources will increase. Although there is no cropland shortage at present (USDA 1992), problems undoubtedly will develop in the near future in response to the diverse needs of the growing US population.

Solar energy technologies, most of which require land for collection and production, will compete with agriculture and forestry in the United States and worldwide (Table 2). Therefore, the availability of land is projected to be a limiting factor in the development of solar energy. In the light of this constraint, an optimistic projection is that the current level of nearly 7 quads of solar energy collected and used annually in the United States could be increased to approximately 37 quads (Ogden and Williams 1989, Pimentel et al. 1984). This higher level represents only 43% of the 86 quads of total energy currently consumed in the United States (Tables 1 and 3). Producing 37 quads with solar technologies would require approximately 173 million ha, or nearly 20% of US land area (Table 3). At present this amount of land is available, but it may become unavailable due to future population growth and increased resource consumption. If land continues to be available, the amounts of solar energy (including hydropower and wind) that could be produced by the year 2050 are projected to be: 5 quads from biomass, 4 quads from hydropower, 8 quads from wind power, 6 quads from solar thermal systems, 6 quads from passive and active solar heating, and 8 quads from photovoltaics (Table 3).

Another possible future energy source is fusion energy (Bartlett 1994, Matare 1989). Fusion uses nuclear particles called neutrons to generate heat in a fusion reactor vessel. Nuclear fusion differs from fission in that the production of energy does not depend on continued mining. However, high costs and serious environmental problems are anticipated (Bartlett 1994). The environmental problems include the production of enormous amounts of heat and radioactive material.

The United States could achieve a secure energy future and a satisfactory standard of living for everyone if the human population were to stabilize at an estimated optimum of 200 million (down from today's 260 million) and conservation measures were to lower per capita energy consumption to about half the present level (Pimentel et al. 1994). However, if the US population doubles in 60 years as is more likely, supplies of energy, food, land, and water will become inadequate, and land, forest, and general environmental degradation will escalate (Pimentel et al. 1994, USBC 1992a).

Fossil energy subsidies should be greatly diminished or withdrawn and the savings should be invested to encourage the development and use of solar energy technologies. This policy would increase the rate of adoption of solar energy technologies and lead to a smooth transition from a fossil fuel economy to one based on solar energy. In addition, the nation that becomes a leader in the development of solar energy technologies is likely to capture the world market for this industry.

Conclusions

This assessment of alternate technologies confirms that solar energy alternatives to fossil fuels have the potential to meet a large portion of future US energy needs, provided that the United States is committed to the development and implementation of solar energy technologies and that energy conservation is practiced. The implementation of solar technologies will also reduce many of the current environmental problems associated with fossil fuel production and use.

An immediate priority is to speed the transition from reliance on nonrenewable energy sources to reliance on renewable, especially solar-based, energy technologies. Various combinations of solar technologies should be developed consistent with the characteristics of different geographic regions, taking into account the land and water available and regional energy needs. Combined, biomass energy and hydroelectric energy in the United States currently provide nearly 7 quads of solar energy, and their output could be increased to provide up to 9 quads by the year 2050. The remaining 28 quads of solar renewable energy needed by 2050 is projected to be produced by wind power, photovoltaics, solar thermal energy, and passive solar heating. These technologies should be able to provide energy without interfering with required food and forest production.

If the United States does not commit itself to the transition from fossil to renewable energy during the next decade or two, the economy and national security will be adversely affected. Starting immediately, it is paramount that US residents must work together to conserve energy, land, water, and biological resources. To ensure a reasonable standard of living in the future, there must be a fair balance between human population density and energy, land, water, and biological resources.

Acknowledgments

We thank the following people for reading an earlier draft of this article, for their many helpful suggestions, and in some cases, for providing additional information: A. Baldwin, Office of Technology Assessment, US Congress; A. A. Bartlett, University of Colorado, Boulder; E. DeMeo, Electric Power Research Institute; H. English, Passive Solar Industries Council; S. L. Frye, Bechtel; M. Giampietro, National Institute of Nutrition, Rome, Italy; J. Goldemberg, Universidade de Sao Paulo, Brazil; C. A. S. Hall, College of Environmental Science and Forestry, SUNY, Syracuse; D. O. Hall, King's College, London, United Kingdom; S. Harris, Oak Harbor, WA; J. Harvey, New York State Energy Research and Development Authority; B. D. Howard, The Alliance to Save Energy; C. V. Kidd, Washington, DC; N. Lenssen, Worldwatch Institute; L. R. Lynd, Dartmouth College; J. M. Nogueira, Universidade de Brasilia, Brazil; M. G. Paoletti, University of Padova, Italy; R. Ristenen, University of Colorado, Boulder; S. Sklar, Solar Energy Industries Association; R. Swisher, American Wind Energy Association; R. W. Tresher, National Renewable Energy Laboratory; L. L. Vant-Hull, University of Houston; Wang Zhaoqian, Zheijan Agri-

cultural University, China; P. B. Weisz, University of Pennsylvania, Philadelphia; Wen Dazhong, Academia Sinica, China; C. J. Winter, Deutsche Forschungsanstalt fur Luft und Raumfahrt, Germany; D. L. Wise, Northeastern University, Boston; and at Cornell University: R. Barker, S. Bukkens, D. Hammer, L. Levitan, S. Linke, and M. Pimentel.

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SOCIO–ECONOMIC AND POLICY ISSUES FOR SUSTAINABLE FARMING SYSTEMS

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Cooperativa Amicizia S.r.l. Padova, Italy (1993)

Printed by:
Cooperativa Amicizia S.r.l.
Via M. Sanmicheli 60, Padova, Italy

Layout:
Aldo Palazzo & Nicola Palazzo

Printed in Italy, December 1993

Environmental and economic benefits of sustainable agriculture

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ABSTRACT

Soil, water, energy and biological resources are being wasted in developed and developing countries throughout the world. In U.S. agriculture, about $44 billion is lost annually because of soil erosion and land degradation. Groundwater is being mined 25% faster than the natural recharge rate and about 1000 litres of oil equivalents are required to produce a hectare of maize. Also, biological diversity, that is essential to a productive agriculture, is being lost due to poor farming practices. This paper describes the use of available, ecologically sound, cultural practices for use in the United States and other developed nations. These will not only maintain high yields but may, in some instances, actually increase yield while reducing production costs and protecting the quality of the environment.

INTRODUCTION

Throughout the world, agriculture is suffering from many serious environmental problems. Soil is being lost from croplands 20 to 40 times faster than it can reform (Pimentel et al., 1993). Each year, about 7 million ha of agricultural land becomes unproductive and is abandoned (Tolba, 1989). Associated with this serious erosion is the rapid runoff of needed water and the loss of nutrients from agricultural lands, that combine to further reduce land productivity (Lal and Stewart, 1990).

Degraded agricultural lands require more fertilizers and more irrigation in order to maintain production (Pimentel and Wen, 1990). This is costly in terms of energy and capital. In addition, abandonment of some technologies, like crop rotations, has resulted in increased insect pests, plant pathogens and weeds, and these in turn have required the intensive use of pesticides (Pimentel et al., 1991). As a result, the costs of agricultural production have escalated.

In addition to the direct effects of poor environmental resource management on agricultural production, the offsite environment has also been seriously damaged. Soil sediments and rapid water runoff from the land are estimated to cause about $6 billion in damages to the environment annually (Clark, 1985). The offsite environmental damage caused by pesticides is estimated to cost at least $2 billion annually.

In order to be able to remedy these serious problems and enable agricultural production to be more economically and environmentally sound in the future, the vital resources required by all agriculture must be identified and their damaging impact understood. In this paper, I examine the diverse environmental problems associated with agriculture and how these increase the costs of crop and livestock production while at the same time causing serious offsite environmental problems. The environmental and economic benefits of sound ecological resource management is also assessed. Although the focus is primarily on U.S. agriculture, the findings can be extended to the environmental and economic problems that currently exist in other nations.

ENVIRONMENTAL PROBLEMS

Soil erosion

World and U.S. food supplies depend on the availability of productive land. Currently, 98% of human food supply comes from the land rather than the oceans and other aquatic systems (Waggoner, 1984). Thus, the dimensions of land destruction in the United States and the world are of increasing concern. At present, soil erosion on U.S. cropland averages about 18 times faster than soil reformation. In Africa, Asia and South America, erosion is 30 to 40 times faster than reformation (Khoshoo and Tejwani, 1993; Lal, 1993; McLaughlin, 1993; Wen, 1993; Pimentel et al., 1993). Because of erosion and other land degradation practices now prevalent in agriculture, as mentioned, about 7 million ha of agricultural land are abandoned each year (Tolba, 1989). This loss is occurring just at the time when more land is needed to grow food for the rapidly growing world population. Finding land and increasing food production to feed the nearly 400 million babies born each day is a sobering reality that we must face (PRB, 1990). In fact, an additional 4 million ha/yr of new cropland has to be accessed and put into production in order to feed these new people. Adding this 4 million ha to the 7 million ha of degraded area means that about 11 million ha of agricultural land has to be secured and put into crop production to maintain the basic food supplies for the people. At present, valuable forest lands are being cleared to provide the "new" agricultural land. As a result, deforestation has reached a crisis status (Pimentel et al., 1986).

Erosion adversely affects crop productivity by reducing the availability of water, nutrients, soil biota, organic matter and soil depth (OTA, 1982). Reduction in the amount of water available to the crop is considered the most harmful effect of erosion (Follett and Stewart, 1985). After water, shortages

of soil nutrients are the most important factors limiting crop productivity. One metric ton of rich agricultural topsoil contains a total of 4 kg of nitrogen as well as other nutrients essential for crop production (Alexander, 1977). Thus, associated with the tremendous amount of topsoil lost in the United States is the loss in fertilizer nutrients that totals $ 18 billion annually (Troch et al., 1980).

Erosion also rapidly removes organic matter from the soil, because it is much lighter in weight than soil mineral elements. Organic matter is important to crop production because it helps retain water, improves soil structure, encourages soil biota and is the source of a large portion of the nutrients needed by crops (Pimentel et al., 1987).

The reason that crop yields have continued to grow during the past 40 years, despite soil degradation, is the 20–fold increase in the application of some fertilizer nutrients, increased irrigation, greater use of pesticides and the introduction of new, high–yielding varieties (Pimentel, 1988). In general, farms have been substituting a non–renewable resource (oil) in the form of fertilizers, pesticides and other inputs for loss of a renewable resource, soil.

In addition to reducing the productivity of the land, erosion and water runoff cause offsite environmental effects. Estimates are that runoff in the United States delivers approximately 3 billion tons of sediment each year to waterways in the 48 contiguous states (NAS, 1974). The sediments have to be dredged from harbours, rivers and reservoirs. Also, these sediments are detrimental to agriculture, burying young crop plants in the low–lands. The heavily sediment–loaded water is a hazard to industrial machinery. Several species of fish, including salmon and trout, are prevented from reproducing when streams and rivers contain heavy sediment deposits. Just the offsite environmental effects of erosion sediments cost an estimated $6 billion annually (Clark, 1985).

These severe soil erosion and associated rapid water runoff problems are now seriously diminishing the food economy as well as the health of the environment. In total, soil erosion and associated runoff cost the U.S. an estimated $44 billion annually in direct and indirect effects (Pimentel et al., 1987).

WASTING WATER

Because all crops require and transpire massive amounts of water, losses in available water inhibit crop growth. Consider a corn crop, which produces about 7000 kg/ha, takes up and transpires about 4.2 million litres of water during the growing season (Leyton, 1983). Although sufficient rain usually

falls upon eastern U.S. agricultural land, periodic droughts limit yields, as was evident during the recent summer of 1988. Elsewhere in the U.S., particularly in the Southwest, the land is arid and must be irrigated to be used for agricultural production. On average, 10 million litres/ha of water are applied to irrigated land each year (Postel, 1989).

Although agriculture pumps about one-third of the total water from streams and aquifers, agriculture consumes 85% of total U.S. water (NAS, 1989). The public, industrial and rural sectors consume the remaining 15%. The reason that agriculture consumes so much water is because of evaporation and transpiration. In contrast, the public and industry use the water but they return it to the stream or lake – it may be polluted but they return the water they use.

Concern about water availability is growing, both because of heavy pollution (Postel, 1989) and excessive overdraft of aquifers (USWRC, 1979). In the 48 contiguous states, water overdraft exceeds replenishment by about 25%, and in the Texas–Gulf area, overdraft is as high as 77%. This mining of aquifers is becoming a major environmental problem because the replacement of ground water is slow, at a rate of less than 1% each year provided that rainfall is adequate (CEQ, 1980).

Special pollution problems associated with irrigated agriculture such as river and stream water as well as the land itself are degraded by the addition of salts. For example, as the Colorado River flows through Grand Valley and water is withdrawn for irrigation and later returned to the river, about 18 t/ha of salt are leached from the irrigated land and added to the detriment of river water (EPA, 1976). At times during the summer, the Red River in Texas and Oklahoma is more saline than the oceans (USWRC, 1979).

Unfortunately, an enormous amount of irrigation water is just wasted in the western states, primarily because the cost is subsidized and farmers essentially have free use of water. For instance, Utah farmers pay only $44/ha for water from the Bonneville Water Project, while the U.S. Government subsidizes this at a cost of nearly $1500/ha (Washington Post, 3/8/88). If farmers were paying the $1500/ha, they would be more careful in the use of this irrigation water. Growing low value forage crops to feed livestock as is done now would be uneconomical.

PESTICIDE PROBLEMS

Each year nearly 500,000 tons of pesticides are applied to U.S. agriculture at a cost of more than $4 billion (Pimentel et al., 1991); world-wide about 2.5 million tons are used at a cost of about $18 billion (Pimentel, 1990a). Despite

this heavy use of pesticides, about 37% and 35% of all crops in the United States and the world, respectively, are lost to pests each year. The crop loss attributed to pests has been increasing slowly for many years in the United States. For example, the share of crops lost to insects has nearly doubled since 1945 even though the use of synthetic insecticides has increased 10-fold (Pimentel et al., 1991). This rise in crop losses, despite increased insecticide use, can be accounted for by the many major changes that have taken place in agricultural technologies over the decades. Some of these changes include reduced crop rotations; greater use of crop monocultures; and the need to adhere to more stringent "cosmetic standards" set by the Food and Drug Administration (FDA).

Of particular concern is that 99.9% of the pesticide that is applied never reaches the target pests, but instead disperses widely to contaminate the environment (Pimentel and Levitan, 1986). Furthermore, pesticides applied by aircraft are wasted because only 25% to 50% of the pesticide ever reaches the target hectare under the most ideal spraying conditions (Mazariegos, 1985; Pimentel and Levitan, 1986). The remainder drifts offsite to contaminate the environment and sometimes threaten the health of people and animals.

For these reasons, the economic benefits of pesticides need to be balanced against waste, as well as the indirect environmental and public health costs, which are estimated to range from $3 to $4 billion each year (Pimentel et al., 1991). Perhaps the most serious social cost is that of human pesticide poisonings. Each year in the United States about 21,000 accidental poisonings are reported, with about 35 fatalities (Blondell, 1989 personal comm). World-wide, the estimate is 1 million human pesticide poisonings with about 20,000 deaths (WHO/UNEP, 1989).

A recent study demonstrated that it would be possible to reduce pesticide use by one-half in the United States, provided a wide array of currently available non-chemical pest management practices were implemented (Pimentel et al., 1991). If this were done, crop yields would remain the same or increase and production costs would increase about $1 billion/yr. This would add only about 0.6% to consumers' food costs. However, if the benefits accrued from reduced environmental and public health risks are subtracted, the actual cost to society of reducing pesticides might be zero or even show a return on the investment in non-chemical controls. Such plans to reduce pesticide use are already under way in Denmark, Sweden, the Netherlands and the Province of Ontario (Pimentel et al., 1991).

LIVESTOCK MANURE

Each year more than 5 billion livestock in the United States produce about 1.6 billion tons of manure (ERAB, 1981). This amount of manure contains more than 5 times the total amount of nitrogen fertilizer that is applied annually as commercial fertilizer to U.S. crops. Half of this manure is deposited in pastures and rangeland and provides some fertilization. Of the half that is collected, about 50% is lost due to poor management practices. For example, approximately 50% of the nitrogen is lost from the manure within 24 hours, if it is not buried immediately or placed in a manure–pond under anaerobic conditions (Exner et al., 1989). As a result, only about 20% of the nutrients in collected livestock manure are captured and available for crop production. Furthermore, a significant portion of the nutrients that are lost in runoff contaminate groundwater and adjacent rivers and lakes (Pimentel, 1989).

FOSSIL ENERGY

To produce a hectare of corn using hand tools requires about 1200 hours of labour (Lewis, 1951). In the U.S. today, a hectare of corn is produced with a farm labour input of about 10 hours (Pimentel and Wen, 1990). The reduction in labour input has been accomplished through farm mechanization that uses enormous amounts of fossil fuel. For instance, to produce a hectare of corn requires about 1100 litres of oil equivalents (Pimentel and Wen, 1990).

Furthermore, fertilizer inputs have increased as much as 20–fold just since 1945 (Pimentel and Wen, 1990). Also, pesticide inputs have risen 33–fold since 1945. Producing fertilizers and pesticides requires a large input of fossil fuels. The recent major changes in agricultural technology have occurred in a relatively short period of time and are consuming significant quantities of fossil energy.

ETHANOL FROM GRAIN

Proponents of producing ethanol from U.S. corn and other grains claim that it reduces oil imports and saves the nation money (ERAB, 1981). Unfortunately, the opposite is true. Each gallon (3.8 litres) of ethanol requires 10.1 kg of corn and costs about $1.94/gal to produce (Pimentel, 1991). Assuming that distillers' dried grains were produced and utilized, this cost could be reduced about as much as $0.60/gal. However, most of this benefit is lost because of environmental pollution that costs at least $0.36/gal. In addition to this cost, federal and state subsidies average $0.79 per gallon (EPA, 1990).

Thus, a gallon of ethanol costs the consumer $2.55 to produce compared with about $0.70/gal of gasoline (Pimentel, 1991).

Also a gallon of ethanol has only about two–thirds as much energy as a gallon of gasoline. To produce a gallon of ethanol in a large 60 million gal/year plant with all modern facilities requires an energy input of 35,046 kcal. A gallon of ethanol contains only 19,450 kcal. This means that it takes about 80% more energy to produce a gallon of ethanol than can be obtained in net fuel. Therefore, not only does the nation have to import oil from the Middle East to fuel this corn/alcohol system but ethanol production is costing the taxpayer huge sums of tax money in the form of subsidies and its production adds to environmental degradation of land, water, energy and biological resources.

Assuming zero energy input for the fermentation and distillation processes of ethanol production and charging only for the fossil energy expenditure to culture the corn (essential to have corn/alcohol produce net energy), the amount of cropland required to fuel just one U.S. automobile is enormous. Making these assumptions, more than 6 hectares of corn land would be necessary to fuel one automobile for one year. In contrast, one person is fed using only 0.6 ha of cropland (USDA, 1989). This emphasizes the tremendous waste of agricultural resources when ethanol is produced from grains.

SUSTAINABLE MANAGEMENT OF NATURAL RESOURCES

The major difficulties associated with conventional, high input agriculture are: (1) high costs of production (Pimentel et al., 1989); (2) serious environmental resource degradation (Pimentel, 1990b); and (3) instability of crop yields (Brown, 1984). Numerous agricultural technologies already exist that can be implemented to make agriculture sustainable and ecologically sound. These technologies would reduce chemical inputs (including commercial fertilizers and pesticides), reduce soil erosion and rapid water runoff, and make better use of livestock manure (NAS, 1989; Paoletti et al., 1989).

The economic and environmentally sound agricultural practice of the ridge–planting and rotation system is compared with the conventional system of producing corn (Tab.1). Note the high level of inputs in the conventional corn system. The total costs of these inputs average $523/ha in the United States, and do not include the average cost of irrigation water. The total energy input is 7.8 million kcal/ha but this would increase to about 11.0 million kcal/ha if the average irrigation input were included (Pimentel and Wen, 1990). The present day yield is about 7000 kg/ha/yr, which is excellent compared with corn yields obtained when 1/20th the fertilizer and 1/33rd the

TABLE 1(*)

Energy and economic inputs per hectare
for conventional and alternative corn production systems.

	Conventional			Ridge-planting & Rotations		
	Qty	10^3 kcal	Economic	Qty	10^3 kcal	Economic
Labour (hrs)	10[a]	7[f]	50[r]	12[cc]	9[f]	60[r]
Machinery (kg)	55[b]	1,485[g]	91[s]	45[dd]	1,215[g]	75[s]
Fuel (litres)	115[b]	1,255[h]	38[t]	70[ee]	764[h]	23[t]
N (kg)	152[b]	3,192[i]	81[u]	(27[t])[ff]	5591[i]	17[mm]
P (kg)	75[b]	473[j]	53[v]	34[gg]	214[j]	17[v]
K (kg)	96[b]	240[k]	26[w]	15[hh]	38[k]	4[w]
Limestone (kg)	426[b]	134[l]	64[x]	426[ii]	134[l]	64[x]
Corn Seeds (kg)	21[b]	520[m]	45[y]	21[b]	520[m]	45[y]
Cover Crop Seeds (kg)	--	--	--	10[jj]	120[jj]	10[nn]
Insecticides (kg)	1.5[c]	150[n]	15[z]	0	0	0
Herbicides (kg)	2[c]	200[n]	20[z]	0[kk]	0	0
Electricity (10^3 kcal)	100[b]	100[o]	8[aa]	100[b]	100[o]	8[aa]
Transport (kg)	322[d]	89[p]	32[bb]	140[d]	39[p]	14[bb]
Total		7,845	$523		3,712	$337
Yield (kg)	7,000[e]	24,746[q]		8,100	29,160	
output/input ratio		3.21			7.86	

amount of pesticide were used. The yield in 1945 with very low inputs was about 1900 kg/ha/yr.

(*) SOURCES TABLE 1:

(a) Labour input was estimated to be 10 hrs because of the extra time required for tillage and cultivation compared with no-till, which required 7 hrs (USDA, 1984a).
(b) Pimentel and Wen, 1990.
(c) Mueller et al., 1985.
(d) Transport of machinery, fuel and nitrogen fertilizer (Pimentel and Wen, 1990).
(e) Three-year running average yield (USDA, 1988).
(f) Food energy consumed per labourer per day was assumed to be 3,500 kcal.
(g) The energy input per kilogram of steel in tools and other machinery was 18,500 kcal (Doering, 1980) plus 46% added input (Fluck and Baird, 1980) for repairs.
(h) Fuel includes a combination of gasoline and diesel. A litre of gasoline and diesel fuel was calculated to contain 10,000 and 11,400 kcal, respectively (Cervinka, 1980).

The environmental costs attributed to both agriculture and society when corn is produced using conventional practices are listed in Tab.2. The loss of fertilizer nutrients totalling $113/ha/yr is based on the calculation of Troeh et

Weighted average value of 10,900 used in calculations. These values include the energy input for mining and refining.
(i) Nitrogen = 21,000 kcal/kg (Dovring and McDowell, 1980).
(j) Phosphorus = 6,300 kcal/kg (Dovring and McDowell,1980).
(k) Potassium = 2,500 kcal/kg (Dovring and McDowell, 1980).
(l) Limestone = 315 kcal/kg (Terhune, 1980)
(m) Hybrid seed = 24,750 kcal/kg (Heichel, 1980).
(n) Energy input for insecticides and herbicides was calculated to be 100,000 kcal/kg (Pimentel, 1980).
(o) Includes energy input required to produce the electricity.
(p) For the goods transported to the farm, an input of 275 kcal/kg was included (Pimentel, 1980).
(q) A kilogram of corn was calculated to have 4,000 kcal.
(r) Labour = $5/hr.
(s) USDA, 1984a.
(t) Litre = $0.33.
(u) N = $0.53.
(v) P = $0.51.
(w) K = $0.27.
(x) Limestone = $0.15.
(y) USDA, 1984a.
(z) Insecticide and herbicide treatments = $10/kg for both the material and application costs.
(aa) kwh = 7c
(bb) Transport = 10c/kg.
(cc) Five additional hours were necessary for collecting and spreading 27 t of manure (Pimentel et al., 1984).
(dd) 20% smaller machinery was used because less power is needed in no-till and ridge planting (Colvin et al., 1982; Muhtar and Rotz, 1982; Allen and Hollingsworth, 1983; Hamlet et al., 1983; USDA, 1984b).
(ee) Nearly 40% less fuel is required compared with conventional because the soil was not tilled, only lightly cultivated (Colvin et al., 1983; Mueller et al., 1985).
(ff) A total of 27 t of cattle manure was applied to provide 152 kg of N.
(gg) A total of 41 kg of P was provided by the manure.
(hh) A total of 81 kg of K was provided by the manure.
(ii) Assumed that same amount of N, P, K, and Ca required in no-till.
(jj) About 10 kg of cover crop seeds were used (Heichel, 1980).
(kk) No herbicide used, weed control carried out by cultivation and cover corp.
(ll) About 1.9 litres of fuel was required to collect and apply 1 t of manure (Pimentel et al., 1984).
(mm) The value of manure was given for the fuel required to transport and spread.
(nn) 1 kg of cover crop seed = $1.

TABLE 2

Environmental costs both onsite and offsite from conventional agriculture per hectare annually (see text for details).

Item	*Costs*
Loss of soil nutrients	$113.00
Loss of water	50.00
Manure pollution	5.00
Sediments impacts offsite	37.50
Pesticide impacts	25.00
Total	$230.50

al. (1980) that $18 billion in nutrients are lost from agriculture via erosion and water runoff annually. The offsite environmental damage caused by sediments is estimated to be $6 billion annually or $37.50/ha of cropland (Clark, 1985). The loss of water caused by rapid runoff was estimated to be a conservative $50/ha (Pimentel et al., 1987). Ground and surface water pollution costs associated with livestock manure were estimated to be a minimum of $5/ha/yr. The yearly environmental costs of pesticides were calculated to be $25/ha based on an estimated $4 billion ecological impact each year from pesticides (Pimentel et al., 1991). Taken together these environmental damages total at least $231/ha/yr for conventional corn production. If these environmental costs were added to the production costs, then the total cost of producing conventional corn rises to $754/ha/yr (Tab.1).

The ridge planting and crop rotation system listed in Tab.1 utilizes readily available agricultural technologies that can make agriculture more productive, economic, sustainable and environmentally sound than conventional corn production. In this system, which uses ridge planting, crop rotations and a cover crop, soil erosion is reduced from approximately 20 t/ha/yr for conventional and continuously grown corn to less than 1 t/ha/yr. Note the 1 t/ha/yr erosion rate equals the soil reformation rate under most agricultural conditions (Hudson, 1981; Lal, 1984 a & b; Elwell, 1985). Also, sound soil and water conservation technologies increase corn yield from 15% to 30% over corn grown under conventional systems that experience moderate to severe soil erosion (Follett and Stewart, 1985; ASAE, 1985). Note that for this analysis I assumed a 15% increase in yield over the conventional corn production system which is a minimum when sound soil and water conservation practices are employed (Tab.1).

Selecting an appropriate crop, like soybeans, for rotation with corn reduces corn rootworm (Pimentel et al., 1993), corn diseases (Pearson, 1976; Mora and Moreno, 1984), and weed problems (NAS, 1968, 1989; Mulvaney and Paul, 1984). Furthermore, a corn and soybean rotation system is more profitable than raising either crop alone (Helmers et al., 1986). In part, this results when corn is grown in rotation because the corn rootworm problem is eliminated and there is no need for insecticide. Average corn loss to insects in conventional, continuous corn is 12%, whereas corn loss to insects for corn grown in rotation is only 3.5% (Pimentel et al., 1991). Thus, corn yields increase more than 8% if insecticides are withdrawn and corn is grown in rotation. For that reason, this 8% was added to the yield in ridge planting and rotation system in this analysis (Tab.1).

Several additional ecologically sound management practices were included in the ridge planting and rotation system (Tab.1). These included recycling livestock manure and use of a cover crop. Using farm manure reduces the pollution of groundwater and/or adjacent waterways, makes use of the valuable nutrients, adds organic matter to the soil, and reduces soil erosion (Pimentel et al., 1987). Cover crops, especially legume cover crops, like winter vetch, reduce soil erosion and water runoff, reduce weed problems and conserve soil nutrients. Soil nutrients are picked up and stored by the cover crop, which is subsequently plowed under to contribute these nutrients once again to the soil.

Ridge planting, the crop rotation and the other techniques included in this particular analysis may not be appropriate for all types of soils, all crops, all pests and all farming systems. However, these technologies were selected for this analysis to illustrate the potential available technologies have to enhance the sustainability of agricultural production. Various combinations of these and other technologies have been developed for particular crops and farming systems (NAS, 1989; Paoletti et al., 1989; Pimentel et al., 1991).

The ridge planting and rotation system has the following advantages over the conventional corn system: 1) soil erosion and rapid water runoff was reduced; 2) smaller tractors were employed and less tractor fuel was used; 3) mechanical cultivation substituted for the herbicides but this was not essential; 4) the rotation eliminated the need for all insecticides; 5) on-farm livestock manure substituted for all the nitrogen and a large portion of the phosphorus and potassium nutrients; and 6) a cover crop protected the soil and nutrients during the non-growing season. The labour input was raised from 10 hours/ha to 12 hours/ha to include the time required to apply the livestock manure to the land.

All these modifications raised the corn yield from 7000 kg/ha in the conventional systems to 8100 kg/ha in the low-input ridge planting and rotation system (Tab.1). Total energy input for the low input system was only 3.7 million kcal or less than half of that of the conventional system. The total cost of production that included the added labour was $337 or 36% lower than the conventional system. If, however, the environmental costs attributed to conventional production had been included, then production costs in the low input system would be about one-half that of the conventional system.

Clearly, the substantially lower production costs of the low input system, plus the 16% higher yield of this system, generate greater profits for the farmer, as well as for society. Specifically soil and water conservation, as well as reduced fertilizer and pesticide inputs, result in major benefits now and in the future.

CONCLUSION

The careless use of soil, water supplies, non-renewable energy, and biological resources is contributing to the current high costs of agricultural production, as well as to the depletion of vital resources in both developing and developed nations. The analysis described in this paper indicates that the use of available, ecologically sound, cultural practices in the U.S. and other developed countries will not only maintain high yields but may, in some instances, actually increase yields while reducing production costs and protecting the quality of the environment.

In addition to ridge planting and rotations, a wide array of soil and water conservation technologies already exist that can be employed in corn and other major crop systems (Troeh et al., 1980; Lockeretz, 1983; Pimentel et al., 1987; NAS, 1989; Paoletti et al., 1989; Pimentel, 1990b). Also the use of non-chemical alternative pest control technologies helps reduce costly pesticide inputs (PSAC, 1965; OTA, 1979; Pimentel et al., 1991).

Selecting the particular combination of alternative practices depends not only on the conditions of soil, water, climate and biota but also on the crop and/or livestock to be produced. Each agroecosystem has to be designed and adapted for the particular biological and socioeconomic environment in both developing and developed nations. In addition to conserving soil and water, the improved use of biological resources for biological control plus obtaining nutrients (nitrogen) from legumes and other technologies can help reduce agricultural production costs.

The proposed ecological approach for sustainable, productive and environmentally sound agriculture calls for more knowledge and a better understanding of the interdependencies of natural resources, crops, livestock and environment than conventional agricultural production has (NAS, 1989; Paoletti et al., 1989; Pimentel et al., 1989). However, this ecological, sustainable approach needs to be implemented because of growing environmental concerns and economic problems plus the challenge of producing more food from the world resources that are available. Clearly, we have more sophisticated ecological knowledge and agricultural technologies than ever before, and future research findings will add to our arsenal of technologies to help agriculture be more productive and more sustainable while at the same time being more environmentally sound.

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