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Sustainability and Agricultural Externalities in Central America at the Intensive Margin: A Critical Review and Synthesis of the Literature

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Bruce A. Larson and José Manuel Perez*

Abstract

This paper aims to interpret empirical evidence in existing literature on environmental and health impacts of agricultural production in Central America. The paper examines those impacts that occur on existing commercialized, agricultural lands, commonly known as externalities at the intensive margin. Concerns center on use of chemicals (mainly pesticides), soil erosion and soil degradation. The links among erosion, soil attributes, degradation, and productivity remain poorly understood and documented, despite years of concern about soil erosion. The literature suggests that soil erosion at the intensive margin does not pose a serious threat to farm productivity. There is severe erosion in Central America, but it is not a significant constraint on future agricultural growth on commercial lands. Farmers make appropriate soil conservation efforts as rational investment and production decisions. The existing literature on health effects of pesticide use remains incomplete, politically sensitive in some places, and difficult to generalize to a national or regional level. The estimated direct costs of just acute pesticide poisonings in Central America are valued annually at 2.5% of total agricultural GDP (about $190 in 1995). Of this total cost, almost 50% is due to suicide-related intentional deaths, 25% is due to accidental deaths, and about 25% is due to non-fatal poisonings.

Keywords: externalities, pesticide poisoning, soil degradation, Central America

JEL codes: Q19, Q24, I18

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"It is very hard to be against sustainability. In fact, the less you know about it, the better it sounds." (Solow, 1993, P. 179)

I. INTRODUCTION

Even though there is little agreement on a practical definition of *sustainability*, most notions of sustainability recognize the need for a clearer understanding of all the benefits and all the costs from various sectors of the economy. Since it is well recognized that all benefits and costs are not adequately reflected in market prices and systems of national accounting, it is widely accepted that a better accounting of externalities is needed. In short, these externalities are the economic benefits and costs *not included in market prices.*[1]

The agricultural sector provides a clear case. Agriculture remains a key sector of most countries in Central America, accounting directly for 14-33 percent of gross domestic product (GDP) (World Bank, 1997), 26-52 percent of the adult labor force (World Bank, 1997), and the majority of exports (see, e.g., USDA, WATI, 1994).[2] The traditional export crops of coffee, bananas, and sugar remain key for the region. While cotton has virtually disappeared, a combination of vegetables, fruits, ornamental plants, and other "non-traditional agricultural exports" have grown rapidly in some countries. Given the links to other sectors of the economy, such as transportation, storage, and food processing, the above numbers understate the overall positive significance of agriculture on the economy at large. The importance of agricultural production for non-marketed home consumption, and as a use for underemployed household labor, is also well recognized.[3] There is widespread concern, however, that past and present agricultural production in Central America has far-reaching environmental impacts that threaten human health, natural ecosystems, and basic economic growth possibilities. For example, Faber

[1] See Appendix 1 for further introduction to the topic of externalities.

[2] For example, as of 1991, agricultural exports accounted for 61% of all exports for Costa Rica, 50% for El Salvador, 66% for Guatemala, 71% for Honduras, and 75% for Nicaragua.

[3] While a clear definition of sustainable is elusive, examples of unsustainable agriculture are easy to find. The short history of cotton production in Central America is a good example. There are clear concerns and some

(1991, p. 33) views existing information as evidence "that the resource base for much of Central America's peasantry has reached a point of ecological collapse." Conroy, Murray, and Rosset (1996, p. 128), based on one unpublished study by Morales et al. (1993), conclude that "survey results from Guatemala suggest that nontraditional crops have seriously undermined the long-term ecological sustainability of agriculture in the highlands through the degradation of soil". Leonard (1987, p. 127) suggests that "the problem of land degradation is reaching crisis proportions in every country of Central America, except Belize", and that over 50% of all land in El Salvador has been serious degraded or faces "serious" erosion. Yet, these direct and indirect negative impacts are not as well understood as the positive direct economic benefits of agriculture.

The use of pesticides is often a focal point in sustainability discussions. For example, Conroy, Murray and Rosset (1996, p. 135) suggest that pesticide poisonings in Central American agricultural zones are among the most frequent and most serious causes of illness in the region. They conclude that pesticide-related health problems in nontraditional export crops in Central America are cause for 'serious concern', and the rate of acute pesticide illness is high. Wesseling, Castillo, and Elinder (1993, p. 234) conclude that "action to stop this epidemic of unnecessary disease and fatalities caused by pesticides in Costa Rica and in many developing countries is urgently needed."

A simple cost-benefit approach to environmental policy suggests that information on both the benefits and the costs is needed in order to evaluate policies designed to protect the environment and human health. The basic conclusion is that environmental and health effects should be reduced as long as the value of the damages avoided (the benefits) are greater than the costs. While such cost-benefit logic is clear, the fact remains that relatively little is known quantitatively about the magnitude of most potential health and environmental impacts of agriculture.

This paper aims mainly to organize, synthesize, and interpret existing empirical evidence on the extent of environmental and health impacts of agricultural production in Central America as a large step in the process of comparing costs and benefits. To achieve this goal, it focuses on impacts that occur at the intensive margin on existing market-oriented, commercialized lands.

reasonable information that suggests at least some of the non-traditional crops may also follow the boom-and-bust path of cotton in the region.

This focus on the intensive margin was chosen because most future agricultural growth will come from these lands. Additionally, hypotheses pose that impacts on these lands will have the greatest implications on long-run issues of sustainable and internationally competitive agricultural production. These on-farm impacts tend to center on chemical use, namely pesticides and fertilizers, and the on-farm impacts of soil erosion and soil degradation, although more recently the positive benefits of different production systems have been recognized, such as shade coffee and migratory bird habitat.[4]

The paper is organized into two main sections. Section II focuses on the health and environmental effects of pesticide use. Section III focuses on the on-farm productivity effects of soil erosion/degradation. Section IV summarizes the main results of these discussions and provides recommendations for agricultural and public health policy, including future data collection and applied, policy-oriented research needs.

II. PUBLIC HEALTH AND AGRICULTURAL USE OF PESTICIDES

While pesticides have played an important role in increasing agricultural growth in Central America, as in the rest of the world, they have also brought negative environmental and health consequences, particularly as indirect consequences, or externalities.[5] Table 1 organizes the possible externalities from agricultural use of pesticides. While these impacts could be organized in several ways, the four categories in Table 1 are based on the way the issues tend to be analyzed and investigated in the health, agricultural, and economics literature.

Analysis of the topic 'pesticides' contains many difficulties that should be acknowledged from the outset. As a group, pesticides include several hundred active ingredients that combine into more than 50,000 commercial formulations as fungicides, rodenticides, insecticides, and herbicides.[6] As an example, Table 3 lists nearly 400 pesticides that are included in the United States Food and Drug Administration's (USFDA) food residue monitoring programs (see FDA, 1996).

[4] Off-site and extensive margin issues are to be addressed in a separate paper by INCAE as part of the Central America Project. These issues are clearly important and include, for example: erosion impacts on dam life, aquaculture, and public water supplies; and negative environmental impacts related to past and future deforestation of primary forests (loss of biodiversity, soil erosion following deforestation and perhaps shifting cultivation, etc.).

[5] See appendix 1 for a short introduction of the topic of externalities.

[6] See, e.g., Castillo, Cruz, and Ruebert (1997) and Murray (1994) for various estimates of total numbers.

Castillo, Cruz, and Ruebert (1997) explain that the main groups of pesticides currently used in Central America are:

(1) organophosphates, carbamates and pyrethroids as insecticides;
(2) dithiocarbamic fungicides; and
(3) phenoxyacids, dipyridyls, and triazines as herbicides. Organophosphates include methamidophos, methyl parathion, terbufos, ethoprophos, and chlorpyrifos. Carbamates include carbofuran, methomyl, and aldicarb (Wesseling, et al, 1997).

Twenty-one pesticides constitute the greatest volume of use in Central America. These are: aldicarb, atrazine, carbofuran, chlorpyrifos, chlorothalonil, 2,4-dichlorophenoxyacetic acid (2,4-D), deltamethrin, diazinon, ethoprophos, glyphosate, malathion, mancozeb, methamidophos, methomyl, methyl bromide, methyl-parathion, paraquat, pendimethalin, propanil, terbufos, and tridemorf.

By their very nature, pesticides are designed to kill, and some are very toxic to a variety of organisms including humans. In Costa Rica, for example, about 30% of all pesticides imported in 1990 were chemicals classified as Category 1A or 1B compounds by the World Health Organization (WHO). These categories designate toxic chemicals to which only trained and licensed application has access in the U.S. and other countries.[7]

1. Acute Health Impacts at the Global Level for Perspective on Central America

Exposure to relatively high doses of pesticides over short periods of time cause acute health impacts. Acute impacts are probably the best known, documented, and understood impacts of pesticide use in the literature. Wesseling (1997) suggests that the "use of toxic pesticides is likely to be one of the most relevant occupational hazards for agricultural workers in the Third World." Jeyaratnam (1990), which summarizes the results of a large World Health Organization (WHO)/United Nations Environment Program (UNEP) project on pesticides and human health,

[7] Pesticides are imported directly and/or formulated and also repacked in about 100 factories in Central America, with most operations located in Guatemala, Costa Rica, Nicaragua, and El Salvador. Most organochlorines such as DDT were banned in the 1970s in the developed world and about ten years later in Central America, although some were being used up to 1986 in Guatemala and through the 1980s in Nicaragua.

provides an accepted, informed guess on the prevalence of pesticide poisonings in the developing world. Jeyaratnam (1990) also suggests that the issue of acute poisonings, because of the magnitude of documented impacts, is the clear priority for developing countries. Low-level chronic impacts are also a concern, but are of lower priority than acute effects.

The acute impacts reported at a global level by Jeyaratnam (1990) provide a good starting point for understanding the possible level of similar impacts in Central America. As of 1985, Jeyaratnam (1990) reports that there are an estimated 3 million pesticide poisonings per year in the world that *lead to hospitalization*. 220,000 of these poisonings result in death.

The Jeyaratnam (1990) study also suggests that the number of hospitalizations would be matched by an equal or greater number of unreported but milder cases of intoxication and other acute conditions such as dermatitis. Using the 3 million figure and using Jeyaratnam's (1990) estimate of 830 million agricultural workers in developing countries, it is estimated that 3/830 = 0.0036, or 0.36% of agricultural workers experience some form of poisoning yearly, half of which are severe enough to lead to hospitalization.

Jeyaratnam (1990) also guesses, however, that perhaps 3% of agricultural workers suffer an episode of pesticide poisoning each year, the majority of which are not reported and officially recorded. This guess is based on the perspective of agricultural workers themselves and review of earlier studies from various Asian countries, including Sri Lanka and Malaysia which are considered to have the best data. In some countries, this number increases to over 7% of agricultural workers per year.

If the 3% estimate is correct, it implies that there could be about 25 million cases of pesticide poisonings in the world annually--about 4 times more than the 6 million estimated also in the same study by Jeyaratnam (1990). The bulk of these cases is mild, not recorded, and does not limit activity. As a result, most workers do not seek medical treatment in such cases. In sum, Jeyaratnam (1990) reports the following annual poisoning rate among the agricultural worker population in developing countries: 0.36% reported poisonings requiring hospitalizations; and 3% total estimate including all types of severe and mild poisonings.

The basic consensus from many countries is that the majority of poisonings, when all types of poisonings are considered, results from suicides and attempted suicides. For example, Jeyaratnam (1990) suggests that about 66% of poisonings globally are from intentional poisonings, and that globally 91% of poisonings leading to deaths are from suicides. Since such issues are not directly related to the agricultural use of pesticides, programs for the safe use of pesticides have little impact of these types of poisonings.[8]

2. Acute Health Impacts in Central America

Wesseling, Castillo, and Elinder (1993) suggest that the rate of poisonings among agricultural workers in Costa Rica is 4.5 per 100 per year. This rate is reasonably consistent with the global estimate of 3% from Jeyaratnam (1990). The 4.5% annual poisoning rate of agricultural workers is based on the observed rate of 1.5% occupational poisonings given medical attention which is then adjust up to account for under reporting.[9] It seems plausible that the Costa Rican rate is higher than the world average due to the relatively large use of pesticides in the region.

For the most part, organophosphates, carbamates, and paraquat are associated with acute pesticide poisonings. Faber (1991) reports that 80% of Central America's pesticide poisonings are from organophosphates, while Wesseling (1997) specifies cholinesterase-inhibiting pesticides (organophosphates and carbamates) and paraquat as causing most of the poisonings (in occupational accident reports, hospitalizations, and deaths).

In an earlier study, McConnell (1988) reports that in Region II of Nicaragua the rate of pesticide poisonings in the general population was estimated at about 2 per 1,000 or about 1,200 cases per year in a population of 632,000. This report, however, does not clearly define poisoning. If the rate its suggests is reasonably correct, then the rate of 0.2% for the total

[8] Van Der Hoek et al. (1998) provide recent detailed evidence on the importance of suicides from pesticides within a rural region of Sri Lanka. Of poisonings reported to hospitals, presumably relatively severe cases, about 73% were due to intentional poisonings and 27% were due to accidental or occupational events. The poisoning rate has increased substantially over the past 20 years to about 43 per 100,000 (430 per million) per year, which is between 4 and 400 times greater than an acceptable additional risk of cancer in the U.S. Van Der Hoek et al. (1998) also suggest that the proportion of 'accidental' poisonings may be overstated in official health statistics because of the social stigma and legal issues associated with suicides. (It is illegal to commit suicide in Sri Lanka.) In 5 of 25 districts in the country, pesticides poisonings were the largest cause of death in government hospitals. For example, in 27 settlements of one agricultural area (district H of the Mahaweli Program) during 1983-1987, suicide was the leading cause of death and 91% of deaths were from pesticides. Van Der Hoek et al. (1998) suggest that suicide in Sri Lanka is driven by impulsive reaction to grief and anger. Easy access to deadly pesticides makes it extremely easy for poorer rural residents to act on these impulses (mainly young men). As a result, the policy issue involves access to pesticides, not use. Programs designed for 'safe-use' are not effective in this context because knowledge of safe use is not the issue.

[9] Wesseling, Castillo, and Elinder (1993) summarize other studies suggesting that the range of under reporting of poisonings could fall in the range of 1:100 (reported: not reported) to 1:6. Wesseling et al. (1997) also summarize unpublished information suggesting that there are over 8 unreported poisonings in one region of Nicaragua for every 1 reported poisoning. There is not a clear understanding of how much "under reporting" is due to mildness, misdiagnosis, lack of physical access to medical care, or costs of medical care.

population could imply a rate of about 0.6% for the agricultural labor force if all poisonings occurred in the labor force. This number is substantially below the 3% number reported above. For reference, about 50% of such poisonings were from the carbamate insecticide Furdan (carbofuran) and about 30% from methamidiphos.[10] Even though this was an area of heavy cotton production in the past, it was estimated that about 80% of all such poisonings occurred in relation to corn production on small- and medium-sized farms (McConnell, 1988). Thus, it may be the case in other parts of Central America that pesticide poisonings are occurring on relatively small farms, not just large commercial areas such as banana production in Costa Rica (whose production has been clearly associated with pesticide health effects).

Following up on the above-reported Nicaragua study, Murray (1994) references Keifer and Pacheco (1991) to suggest that the above information involves substantial under reporting. As a result, Murray (1994) suggests that a better calculation would be in the range of 1.5-3% of the rural population, which is more consistent with the world estimates by Jeyaratnam (1990).

Agne (1996) summarizes information on poisonings in Costa Rica reported to the country's National Center for Poisoning Control located at the Children's Hospital in San Jose. These poisonings should be considered "severe" as the person was sick enough to seek medical treatment and as local doctors could determine that the sickness was related to pesticides. From 1980 to 1994, the cases reported annually increased from 593 in 1980 to 1144 in 1994. Of the 1144 reported poisonings in 1994, 233 were from organophosphates, 151 from carbamates, 128 from paraquat, and the responsible pesticide was not identified in the remaining majority of cases. Strikingly, 34% were classified as 'accidental' poisonings, 43% as occupational, and only 19% as attempted suicides. This attempted suicide rate of 19% in Costa Rica is substantially different from the global estimate of 66% reported by Jeyaratnam (1990).[11]

Agne (1996) also notes an additional feature of the poisoning data; namely that children (under 15 years of age) account for almost 30% of all reported poisonings and over 20% of all poisonings occurred in children under the age of 5 years. Of the 1082 reported poisonings in 1993, there were 43 deaths. This rate of 43 per 1082, or 4% of deaths from reported poisonings, is basically consistent with the 3% rate estimated in Jeyaratnam. Costa Rica uses a relatively

[10]Methamidophos was reported to be among the top three causes of pesticide poisonings annual in each country of Central America (Pesticide Program, 1991).

[11] Of course, this difference could be due to several factors, including lower suicide rates or under reporting of such incidents because of social and religious ramifications.

high quantity of pesticides. It is a relatively high income 'developing country' and has an established poison control system that includes monitoring mechanisms. Thus, it is not surprising that the reported death rate from poisonings is higher than average.

Based on small samples of farmers (number between 25-74), farmers self-reported pesticide poisonings at a rate of 28% over two years in El Salvador and Honduras, 56% in Costa Rica, and 57% in Guatemala (Conroy, Murray, and Rosset, 1996, p. 139). These numbers are substantially higher than reported elsewhere, even accounting for their two-year time period.

3. Possible Levels of Economic Costs Associated with Acute Poisonings

In the literature, 'poisoning' remains unclearly defined or identified in many instances. This creates the main challenge in reviewing this literature from a policy perspective. Difficulties associated with compiling poisoning data, especially misdiagnosis and under reporting, aggravate this challenge. As mentioned above, the concept of 'poison' can range from death to very mild effects that do not restrict activity. Morgan (1989) describes common symptoms from poisonings including salivation, sweating, vomiting, miosis, headache, and light-headedness. These effects can last for a few minutes or hours or substantially longer. In some cases, activity is not restricted, while in other cases it may take days or even weeks to return to normal activity. As a result, the concept of "poisoning" is not that useful for policy purposes. In the future it will be necessary to focus on more detailed symptoms and impacts such as salivation, sweating, vomiting, miosis, headache, light headedness, loss of appetite, lost work time, lost leisure time, etc.

The costs to the economy from acute poisonings are not directly included in profits and domestic production calculations of the agricultural sector. The data on poisoning rates cannot be easily compared to the value of agricultural production and agricultural wages. Thus, it is unclear if the costs associated with these acute poisonings are somehow "large" or "small" relative to the benefits of pesticide use in agriculture.

Table 3.A. and 3.B. attempt such a comparison. They provide first estimates of the economic costs of acute pesticide poisonings in Central America using existing available information.

Rates of Poisonings

Table 3.A focuses on costs associated with non-fatal poisonings in Central America based on a range of reasonable poisoning rate estimates. A rate of 3% is based on Jeyaratnam (1990) and 4.5% on Wesseling, Castillo, and Elinder (1993) from Costa Rican data. A rate of 8% is a higher end estimate meant to include the possibility of substantial under-reporting.

With about 4.42 million people in the agricultural labor force (according to international statistics as presented in Table 2), the 3% rate implies about 132,600 annual cases of acute pesticide-related health effects in Central America. Extending the Costa Rican rate of 4.5% to the whole region, this number rises to 198,900. A rate of 8% increases the number even more, to 352,000 poisonings per year.

Based on the existing literature, the majority of these cases is mild and does not limit activity. Using Jeyaratnam's (1990) study, only an estimated 12% of poisonings are estimated to be serious enough to require hospitalization (3 out of 25 million). If only these health effects were considered, the 'serious' rate would fall from 3% per year to 0.36% per year. As reported above, Wesseling, Castillo, and Elinder (1993), using Costa Rican data, estimate an occupational rate of 1.5% for poisoning serious enough to seek and to receive medical attention. Actual hospitalization occurs at a much lower rate.

Yet, there is a clear probability that poisonings are under-reported because health care is not received, the poisoning is not correctly diagnosed, or the poisoning is relatively mild. In sum, poisoning rates of 3-8% per year for agricultural labor reported in Table 3.A fall within the range of the reasonable, with the majority of poisonings occurring in the milder end of the spectrum.

Costs of Poisonings to the Individual

Given these possible poisoning rates, the next step is to consider the costs to the individual and to the economy of such poisonings. A standard economic approach for valuing health effects evaluates the total willingness to pay to avoid some risk of an unwanted health outcome.[12] Depending on the situation, this valuation exercise could focus on valuing a known health outcome (i.e. severely poisoned and in the hospital for 4 days) or some changed probability of receiving a known health outcome (i.e. a 10% probability of being severely poisoned and in the

[12] For general discussions, see Harrington and Portney (1987), Freeman (1993). For specific examples related to air pollution and health, see Alberini, et al. (1996).

hospital for 4 days). To date, this research encountered no studies that attempted to estimate the willingness of agricultural workers to pay to avoid risks of pesticide poisonings in Central America.[13]

Total willingness to pay to avoid a known health outcome can be separated into four terms (Harrington and Portney, 1987):

(1) lost wages;

(2) *actual* expenses on medical care by the agricultural worker;

(3) *actual* expenses on avoiding the effect in the first place; and

(4) the ‘disutility’ or value of ‘pain and suffering’ from the event.[14]

While the economic logic behind such a calculation is clear, the resulting distributional implications applying this logic to developing countries may cause reasonable concern. For example, suppose that the benefits of pesticide use go to a relatively wealthy land-owning family or corporation, while relatively poor field workers receive the external costs. In such a case, given the low income levels of the workers, any estimates of worker willingness to pay to avoid health effects are likely to be small compared to the benefits of such pesticide use.

As a result, calculations of willingness to pay to avoid a health risk need to be based on the following assumptions, which are often not recognized or forgotten: the entity creating the risk has the right to create the risk; and the existing distribution of income is considered acceptable. If the chain of logic is carried far enough for policy purposes, it is necessary to know the willingness of the victim to pay in order to determine if the victim could compensate a landowner (in this case to reduce the health effects of pesticide use). In short, while willingness to pay is a valid concept for thinking about many types of environmental valuation questions in both developed and developing countries, common sense suggests that caution should be exercised when very unequal wealth situations are being considered.[15] [16]

[13] Undertaking such applied research in Central America would be very useful.

[14] Note that item (2) is actual expenses on medical care, which is not necessarily the same as the expenses needed for the fully appropriate medical treatment as defined by a doctor or health care provider.

[15] It should be emphasized that there are many situations where willingness to pay is a very reasonable approach for estimating economic costs and benefits in developing country contexts. Examples include the provision of local water infrastructure that needs to be self-financed by the local community or the estimation of the local market values of non-timber resources in tropical forests.

[16] In such circumstances, it could be more reasonable to consider the question of willingness to accept such risks. Such an approach may be reasonable for high income risky professions in developing countries (wage differentials for risky jobs), but probably still has the same problems as willingness to pay when transferred to a low income situation in which workers perceive few alternative employment options.

More complete and detailed analysis of the economic costs of acute health effects of pesticide use in Central America are merited, as are studies on the four components of total willingness to pay to avoid health risks. Similar analysis elsewhere in the world, however, provides a starting point to think about the magnitude of these costs.

For example, there are estimates of willingness to pay in the United States to avoid pesticide health risks that could be loosely described as 'severe', 'moderate', and 'mild.' Of severe risks, Viscusi and Magat (1987) report an implied value of $650,000 associated with willingness to pay to avoid chloramine gassing from bleach among a population with an average income of $40,000.[17] Such an event could be compared to a severe pesticide poisoning. If simply extrapolated to Central America where average per-capita GNP is in the range of $1400, the value of an equally severe poisoning could be $650,000*(1400/40000) = $22,000.[18]

On the moderate end, Viscusi, Magat, and Huber (1987) report a willingness to pay $1,504 to avoid a skin poisoning from an insecticide among a population sample with an average gross income of $42,700. In other words, the value of the health outcome (skin poisoning) was 3.5% of annual income. Using the per-capita GNP estimates of 1995 from Table 2 and a willingness to pay based on equivalent percent of income, the willingness to pay to avoid a pesticide skin poisoning would be $13 in Nicaragua and $91 in Costa Rica ($7-$200 using PPP GNP figures). The average for the region would be about $45.

Also on the moderate end, Viscusi, Magat, and Forrest (1988) report a willingness to pay to avoid an inhalation skin-poisoning at about $2,500 among a population with an average income of about $44,500. This implies the health outcome was valued at 5.6% of annual income. By the same method as above (equivalent percent of income) the willingness to pay to avoid an inhalation skin poisoning from a pesticide is $21 in Nicaragua and $146 in Costa Rica ($11-$300 using PPP GNP figures.) The regional average is about $75.

For milder poisonings, Berger et al. (1987) report the following willingness to pay to avoid one day of various illnesses, including: $98 for coughing spells; $63 for itching eyes; $183 for heavy drowsiness; $140 for headaches; and $62 for nausea. These symptoms are fairly similar to some of the effects of mild pesticide poisonings in Central America. While average income is

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[17] While this figure may seem high, a value of $1.60 million was estimated for drain opener hand burns!

[18] This simple reduction in willingness to pay based on income ratios assumes that the income elasticity of demand for risk reduction equals one. If the income elasticity of demand was less than one, which is usually the

not reported in the Berger et al. (1987) study, $40,000 is probably a reasonable guess based on income levels reported from similar types of studies in similar years (see, e.g., table 7 in Viscusi, 1993). Using this income figure, a day of heavy drowsiness of about $183 equals 0.45% of income. This implies a willingness to pay of about $6 for someone with a $1400 annual income.[19]

From the above calculations, an average value for all poisonings in Central America can be calculated. Assume that a severe poisoning is valued in the range of $20,000, a moderate poisoning at about $50, and a mild poisoning at about $6 on average across Central America. If 1% of poisonings were defined as severe, 25% as moderate, and 75% as mild, then a rough average value of a pesticide poisoning would be $217. Thus, as a round number, $200 as a cost on average per 'poisoning' in Central America is reasonable. Readers are encouraged to revise the assumptions in the calculations based on their private opinions to understand the sensitivity of the results to various assumptions.

Costs of Poisonings to the Economy

To continue with Table 3.A, if an average poisoning event (excluding deaths) was valued at $200, then the cost to the economy from the poisoning rates of 3%, 4.5%, and 8% would be about $26, $40, and $70 million, respectively. Excluding intentional poisonings and using the rate of 24% reported for Costa Rica (Wessling, 1997) which is substantially lower than the global estimate, the cost estimate falls somewhat to $20, $30, and $53 million annually. Relative to agricultural GDP, these calculated costs of non-intentional poisonings (but not deaths) could easily be in the range of 0.25% to 0.68% of agricultural GDP. Again, these

case or is assumed for many types of goods, then the willingness to pay of the poorer population (i.e. Central America) would be substantially higher than used in the calculation in Table 3.

[19] It could also be possible to consider a 'poisoning' day, which in some sense resembles a restricted activity day (a RAD)--a day during which normal activity is curtailed (Alberini et al., 1996, p. 115). In Taiwan, with per capita income in the range of $7,000, Alberini et al. (1996) estimated that the willingness to pay to avoid a one day cold that did not require a doctor's visit but led to a RAD was about 24$. Thus, a one-day unwanted health event leading to a RAD was valued at $24 when per capita income on average is $7,000 per year (not PPP equivalent). Given that Costa Rica's per capita income level was $2610 in 1995, it could be estimated that an equivalent willingness to pay in Costa Rica would be about $9.6. This number is somewhat consistent with estimates for mild effects reported in the text.

estimates are relevant just for the acute morbidity effects of pesticide poisonings excluding any other lingering chronic effects as well as deaths.[20]

In sum, the example outlined in Table 3.A identifies the type of additional information that is needed to begin to evaluate the economic costs of acute pesticide poisonings in Central America. It is clear that more applied research on the economic costs of acute pesticide poisonings is needed in Central America to complement the existing and on-going epidemiological data collection and analysis. Direct studies on willingness to pay to reduce/avoid risks of some of the effects of more mild poisonings are a good place to begin.[21]

While the above discussion focuses on acute morbidity effects of pesticide poisonings, it is also possible to follow a similar process with respect to deaths. Regarding estimated deaths associated with pesticide poisonings, Wesseling, Castillo, and Elinder (1993, 234) reported "over 60 deaths" from pesticide poisonings every year in Costa Rica, of which about 62% are reported to have been suicides. Since Costa Rica uses substantially more pesticides (on both per hectare and per capita basis) than other countries in the region, it is likely that rates in the other countries in the region are not higher than the Costa Rican rates.[22] Table 3.B., presents the Costa Rica poisoning mortality rate for the agricultural labor force with an estimate of the agricultural labor force in Central America. From this, it can be estimated that about 1020 deaths from poisonings occur each year in the region, with about 388 from non-intentional poisonings.[23]

From this estimate Viscusi (1993, p. 1942) concludes that "the appropriate measure of value of life from the standpoint of government policy is society's willingness to pay for risk reduction" (Viscusi, 1993, p. 1942). If in a particular country people are willing to pay on

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[20] These costs are estimates of the area B+B'+F+F'+G in Figure 1 just for acute health effects of poisonings not leading to deaths.

[21] For additional reference on the magnitude of acute poisonings in Central America, Table 3.A also reports information on malaria risks in Central America. For example, Table 3.A estimates between 132,000 - 352,000 pesticide poisonings per year in the region. Using information from the Global Health for All Database of the WHO, about 140,000 cases of malaria were reported in Central America in 1996. While these reported malaria cases probably also involve some amount of under reporting, it seems reasonable to conclude that the number of pesticide poisonings per year in Central America are *in absolute number* very comparable to reported malaria cases. Of course to evaluate more fully the risks of pesticide poisoning as compared to the risks of malaria infection, it is necessary to have better information on the relative severity of the different events as well as the costs of reducing both pesticide and malaria health risks.

[22] There is no indication that pesticides used in the other countries are relatively more toxic or are applied in a less-safe manner than in Costa Rica. As a result, given the larger quantities used in Costa Rica, it is hard to argue that deaths rates would be higher in the other countries. Estimates of country-specific deaths besides Costa Rica have not yet been found in the literature.

average $1000 for a 1 in 100 reduction in risk of death, then the implied 'value of statistical life' equals $1000/0.01 or $ 100,000. This approach was established in the 1970s and Viscusi (1993) summaries the topic and a wide range of empirical studies from developed countries on the topic. Viscusi (1993, p. 1942) concludes "most reasonable estimates of the value of life are clustered in the $3 million -$7 million range".[24]

With annual average GNP in the region at about 5% of U.S. levels in 1995, estimates from developed countries could imply a value of statistical life (VOSL) of $150,000 to $350,000 in Central America. Using the lower end VSOL amount of $150,000, deaths from pesticide poisonings incur about $153 million in costs on the economy. Of this, $95 million are attributed to suicides and about $58 are due to non-intentional poisonings. These number more than double, of course, if the higher end estimate of $350,000 was used.

In sum, the lower estimate for VOSL with respect to deaths and the Costa Rican poisoning rate (non deaths), the acute health effects from pesticide poisonings in Central America, and currently available information (see Tables 3.A. and 3.B.), suggest the following costs of acute poisonings in Central America:

Total Cost of Acute Poisonings	**=**	***$193 million***
of which:		
Deaths from Suicides	=	$95 million
'Accidental' deaths	=	$58 million
'Accidental' non-fatal poisonings	=	$30 million
Attempted Suicides, non-fatal	=	$10 million

In total, these figures of just acute human health effects of current pesticide use could easily represent 2% or more of annual agricultural GDP in the region.

[23] Based on occupational accident death rates as reported by the WHO, there are about 1790 occupational deaths and 3648 traffic accident deaths in Central America each year. Thus, total deaths from pesticide poisonings as compared to occupational deaths are large in relative terms. Traffic deaths are substantially larger.

[24] See Freeman (1993) and Viscusi (1993) for clear discussions of the logic behind this approach. It is important to remember that the VOSL approach actually values average willingness to pay for small changes in risk, say a 1% increase in an annual death probability, and then adds up the average payment 100 times. As a result, a willingness to pay $1000 to avoid a 1 in 100 risk of death would imply a VOSL of $1000*100 or $100,000. As a result, VOSL estimates are almost always substantially larger than annual average GNP or other definitions of income. This same logic is applied to morbidity events (e.g. non-death poisonings) discussed in Table 3.A, in which case the implied value can be several times larger than daily wages.

Policy Questions and Issues

While these estimates provide one snapshot on a segment of the direct economic costs of pesticide poisonings in Central America, they do not provide clear guidance on what if anything should be done to reduce such costs. The calculations in Table 3 indicate some benefits that would result from the reduction of the risks of pesticide poisonings in Central America. For example, when attempting to estimate health benefits of programs to reduce poisoning risks, a rough estimate of $150,000 could be used for deaths, $20,000 for severe poisonings, $50 for moderate, $6 for mild poisonings. With these rough benefits estimates, the next step would be to estimate the costs of reducing the various types of risks. Even without these calculations, however, it is clear that whatever relatively low-cost policy and program measures exist that reduce deaths and severe poisonings are warranted.

4. Chronic Health Impacts

Chronic health effects endure for a longer time than acute effects. There is often no clear line between the two types; acute, sub-chronic, and chronic refer to ranges in a continuous spectrum of events. Acute health impacts are probably the best known, documented, and understood impacts of pesticide use, while information on chronic impacts of pesticide use is less available and more difficult to evaluate. Jeyaratnam (1990) also suggests that acute poisonings, because of the magnitude of impacts, are clearly a priority issue for developing countries. The implication is that chronic impacts are also a concern, but of lower priority relative to acute effects.

Jeyaratnam (1990) reports on a global basis about 735,000 annual cases each year of specific chronic impacts from long-term exposures, with an additional 37,000 cases a year due to unspecific chronic effects (e.g. cancers). Jeyaratnam also estimates that there are about 830 million agricultural workers in the world. The annual cases (735,000) divided by number of workers worldwide equals 0.00088—or an implied rate of about 1 per 1000 agricultural workers annually experiencing some form of chronic health effects from pesticide exposures.

These numbers are very uncertain for many reasons, but two stand out. First, the links (dose-response relationships) between pesticide exposures and chronic health effects are not well known for many pesticides. Second, even if the links are known, medical diagnoses may not correctly identify the underlying cause.

5. Chronic Effects from Past Pesticide Use

There are many different types of potential chronic health impacts, with some important portion of chronic health impacts deriving from the past use of pesticides that are no longer used and/or banned in the region. For example, Faber (1991) reported that 700,000 people in Central America have more DDT (an organochlorine) in their body fat than any other population in the world, with 19 of the 25 organochlorines defined as carcinogenic substances. Given relatively recent exposures, impacts from past pesticide use on future cancer rates in Central America are not known at this time.

Jenkins (1995) and Murray (1994) report on research from the early 1970s that documents the presence of DDT in breast milk of Guatemalan women at levels up to 250 times higher than tolerance limits defined by the WHO for DDT in cow's milk. (This is the maximum found in a sample, not a mean or median level.) Swezey et al (1986, p. 29, also see Murray 1994, p. 48) report that women in the city of Leon, Nicaragua had average breast-milk samples of 2.12 ppm, which was nearly 40 times higher than WHO tolerances for DDT in cow's milk. Jenkins (1995) also summarizes earlier research by Espinoza and Thiel (1987) documenting average high DDT levels (600 times above U.S. standards for cow's milk and 240 times similar FAO standards) among a small sample of Panamanian women.

Over the past twenty years, various studies have also found DDT, its metabolites, and other organochlorine insecticides such as toxaphen in soil, plants, vegetables, fish, clams, red meats, cows milk, human fatty tissue, and mother's milk (Jenkins, 1995). For example, McConnell (1988) reports a case near a crop dusting airport where nursing mothers living in the vicinity have levels of toxaphene, a restricted pesticide in the U.S. at the time and since, that was 68 times the U.S. allowable daily intake.

It is not necessary to document all the past studies on organochlorine compounds to conclude that the citizens of Central America continue to live with chronic exposures from pesticides used in the past but currently banned and not used. The level of risk and health effects that occurred in the past and will occur in the future remains very unclear. To a very large degree, however, these health issues are not directly linked to current and future pesticide use in the region except as continuing reminders of the potential long-term consequences of possibly poor and over use of pesticides.

6. Chronic Effects from Current Pesticide Use

Chronic effects from currently used pesticides in Central America also exist. McConnell, Keifer, Rosenstock (1994) begin to document the chronic effects that follow from acute poisonings of currently used pesticides (e.g., organophosphates). Cherniack (1988) notes that many organophosphates available on the market were never tested for more chronic, neurotoxicological effects because they were on the market before such testing was required. Thus to some degree, since there was a presumption that chronic effects did not occur, data was not gathered on the topic for some time. This highlights the point made earlier that the links between pesticide exposures and chronic health effects are not all that clear given existing scientific knowledge. The fact that such effects are not well documented does not imply they are not real, just that science is not developed enough to understand the likely and actual effects.

As just one example of science following commercialization, chlordimeform came on the market in the 1960s as an effective pesticide (an ovicide controlling pests while still in the egg or larval stage) for bollworm control in cotton, although it was also effective for other insects. It was less persistent than organochlorines and less acutely toxic than organophosphates. Its carcinogenic properties, however, were reported in 1976, and the chemical was initially withdrawn from the market before returning to the market to become a major product for Ciba Geigy. (It constituted about 5% of annual sales in mid-1980s or about $180 million annually.) Murray (1994) cites calculations by Hooper (1982) to suggest that, using a quantitative risk assessment methodology of the California Department of Health Services, workers with high exposure to chlordimeform (5 days a week for 3 months a year for 20 years) would have an expected increased risk of cancer of 1/1000. This is 100 to 1000 times greater than considered acceptable in the U.S. Nicaragua suspended registration of the chemical for use in 1987, a decade after its carcinogenic properties were reported.[25]

Besides concerns such as cancer, many other types of chronic health effects are documented in the literature. For example, Faber (1991) reports that more than 1,000 male workers from the Atlantic banana-growing region of Costa Rica were sterilized from exposure to the nematicide DBCP, with another 5,000 workers at “risk”. Wesseling et al. (1997) also report that the rate of contact dermatitis caused by pesticides was about 1% per year among banana plantation workers. At this time, it is difficult to hazard a guess about the magnitude of chronic health impacts from *current* pesticide use in Central America. The presumption in the public health literature seems to be that the health effects in total are less pressing than acute effects. At the same time, this presumption may be biased by the lack of understanding of chronic health impacts from pesticide use.[26]

7. Residues in Natural Environment and Food

While pesticide residues in the natural environment and food may not technically be considered an impact that occurs on the farm, these residues exist in the rural environment and may related to additional rural health impacts from agricultural pesticides. As a result, the topic is included briefly here.

Pesticide residues exist in the environment and food in a large number of countries. Table 5 shows the 20 different pesticides that were found in more than 2% of samples from representative market baskets in the United States (based on data from 1996). Most notably, DDT was found in 18% of all samples, with malathion, chlorpyrifos, endosulfan, and dieldrin also found in 10% or more of the samples. The frequency of occurrence does not imply that the levels are high, but does confirm that persistent compounds such as DDT continue to show up in foods long after their use is banned.

For lower income countries, Albert (1996) emphasizes that the data on pesticide residues in the environment are quite scarce and difficult to obtain. This is clearly the case in Central

[25] Murray (1994) suggests that several hundred thousand people were exposed to this pesticide during its twenty year life on the market. Using estimated increased cancer risk of 1/1000 also cited, the implication is that perhaps several hundred additional cases might occur over several decades from the use of just this one chemical.

America, although there is a growing literature, much of it unpublished, on site-specific monitoring studies. For example, Castillo, Cruz, and Ruepert (1997) summarize the results of 17 studies of pesticides in the aquatic environment in Central America ranging from 1970 in Guatemala to 1995 in Costa Rica. Chemicals commonly found in surface waters, sediment, and aquatic life in the studies included unspecified organochlorines, toxaphene, parathion, methyl parathion, propanil, unspecified organophosphates, paraquat, PCBs, aldrin, dieldrin, endrin, chlorothalonil, chlorpyrifos, methamidofos, and the like.

Kammerbauer and Moncada (1998) report information on water samples from three locations in Honduras (La Lima, Zamorano, and Choluteca) associated with different agricultural production systems. Samples from soils, rivers, lagoons and wells included detectable levels of heptachlor, endosulfan, and chlorpyrifos. Ten pesticides, mainly organochlorines, were found in fish samples. Pesticides were detected in 20% of samples. Also, spring water in an area of 'traditional' agricultural production included detectable levels of organochlorines and chlorpyrifos during the rainy season. The authors conclude that current levels of pesticides in the ambient environments studied did not pose serious threats to human health but such issues should continue to be monitored in the future. For example, chlordane (total) was detected at levels of 0.25 mg/kg in the Choluteca river, dieldrin reached concentrations of about 0.04 mg/kg, endosulfan concentrations were found in the range of 0.06mg/kg, and total parathion concentrations were about 0.1 mg/kg in the same river.

Meyer (1998) provides recent information on samples from aquatic environments in Honduras during 1995-1998. Based on a total of 80 water samples, organochlorine pesticides are commonly found in samples of coastal and inland waters. While most samples were at low levels (less than 0.010 ppm), there were some samples from the Choluteca and Nacaome rivers at rather high levels (e.g. 2.8 ppm diallate, 9.23 ppm Carbofuran, and of melons and other export crops occurs 1.79 ppm Propiconizole). It is noted that intensive production of melons and other export crops occurs in this region. This study estimated LC50 levels of endosulfan and lindane that were fairly low for post larval shrimp.[27]

[26] Besides these direct effects of pesticide exposures on human health, there are also concerns that some pesticides may impair human and animal immune systems, thereby weakening responses to various infectious and parasitic diseases (World Resources Institute, 1996).

[27] Meyer (1996) reports that pesticide contamination of coastal waters is implicated in mass mortality of cultured shrimp with significant financial losses for farmers in Ecuador and areas in Central America.

Castillo, Ruepert, and Solis (1998) analyzed 111 samples of surface waters (drainage canals and creeks, the Suerte River, and the Suerte River mouth) from the Rio Suerte Basin during 1993 and 1994. This area is a banana-growing region that drains into the Tortuguero Nature Conservation Area on the Atlantic Coast of Costa Rica. This study, which included 11 of 21 commonly used pesticides in banana production in the region, found detectable levels of 10 of the 11 pesticides in the environment. For example, cadusafos levels in the ambient environment of drainage canals and creeks, the Suerte River, and the Suerte River Mouth were on average 0.16 micrograms per liter in all positive samples. (The 69% positive rate had a range of 0.02 to 0.4 micrograms per liter.) This implies an average rate of 0.11 micrograms per liter for all samples in the river. A toxic reference value for this pesticide in water is 0.16 micrograms per liter. The pesticide propiconazole was also found in 50% of river samples with an average value of 0.22 microgram per liter for positive samples for an overall average of about 0.11 micrograms per liter. A toxicity reference value for this compound is 0.20 micrograms per liter.

Regarding pesticide residues on foods, Jenkins (1995) also summarizes a recent study by Rodriguez and Lamoth (1994), where 22% of 229 samples from 32 different crops contained pesticide residues. Most levels were above international norms. Consistent with recent sampling data from the U.S. (see Table 5), the pesticides that were detected include DDT, dieldrin and others that persist in the environment but are no longer legally used in the countries.

While some sampling data exists and additional studies are likely to be completed in the near future, such sampling activities have to be organized in a way so that results can be made more clearly useful in public policy discussions. Statistically useful sample sizes (large enough numbers and time distributions) and summary sample statistics are clearly needed in order to understand more fully levels in the ambient environment.

8. Pesticide Residues, Detentions, and Loss of Export Revenues

As summarized in Table 1, pesticide use can lead to direct economic losses, as opposed to indirect losses due to human health discussed about. Export losses due to detentions at the border of foreign markets (i.e. the U.S.) eventually affect on-farm profits. Economic costs also result from off-site damages to other sectors of the domestic economy (pesticides in surface waters damaging aquaculture). The remainder of this section focuses on the economic impact of detentions.[28]

During the 1980s and early 1990s, a time of large growth of non-traditional agricultural exports crops from Central America to the U.S., several factors led to a growing awareness of the potential for export losses due to pesticide residues on mainly fresh food products destined for external markets. Concerns in the U.S. about the 'circle of poison' led to better data and information on pesticide residues on fresh food products in the U.S. Information and lobbying caused highly publicized problems for Chilean grapes and U.S. apples.

Better information on detentions has resulted in initial calculations on the magnitude of the export losses. For example, Thrupp (1995) suggests that U.S. FDA detentions for pesticide shipments in fruits and vegetables imported to the U.S. from Costa Rica, El Salvador, Guatemala, and Honduras were 'valued' at about \$19.5 million in total over the years 1984-1994. About \$16 million came from Guatemala alone during 1992-1994, with \$10 million accruing in 1993 alone (Thrupp, 1995, p. 157).

While Thrupp's analysis (1995) does not make clear how the 'value' was determined (FOB, CIF, retail), it is surprising how *small* these estimates are. While the detentions over 10 years are valued at about \$20 million in nominal terms, annual agricultural GDP in the region (excluding Nicaragua because Thrupp (1995) reported no detentions for the country) was about \$7.9 billion in 1995. This suggests that the detentions over 10 years constitute less than 0.3% of just one year's agricultural GDP.[29]

Barham et al (1992, p. 68) report that non-traditional agricultural exports were about \$520 million in 1990 in Costa Rica alone. Thus, relative to just Costa Rica's exports of non-traditional (e.g., excluding coffee, bananas, etc.), the Thrupp (1995) detention estimates for Costa Rica of \$411,000 over ten years represents about 0.08% of a portion of Costa Rica's agricultural exports for just one year. Since farm-gate prices are only a portion of FOB prices (perhaps 20-40% depending on crop and country) not all of these detention losses are paid by farmers. Instead some losses are probably borne by groups higher in the marketing chain.

[28] As early as 1970 in Guatemala, it was recognized that pesticide applications in cotton fields were having a negative impact on the shrimp industry. Meyer (1996) also documents such issues in relation to Honduras.

[29] Thrupp (1993) also mentions that the majority of such detentions are released for entry into the U.S.. In reference to Guatemala's problems, Thrupp (1993) indicates that exporters and the U.S. FDA relatively quickly organized to manage the country's problems. Thus, while Guatemala faced the main problem with export detentions

Thus, from information currently available, there is little indication that lost export revenues from pesticide detentions is fundamentally a serious problem for exports from Central America as a whole. Guatemala remains the exception. Various types of peas from Guatemala continue to be placed on an automatic detention without physical examination by the FDA (see attachment A - Import Alert #99-15 12/2/96) for chlorothalonil and methamidophos. While 3 Guatemalan shippers/manufacturers of frozen snow peas have received exemptions from the automatic detention policy, most of Guatemala's U.S. detentions (54 of 57 detentions) during July 1998 were for fresh snowpeas.[30]

There continue to be a few shipment detentions each month from the other countries, such as: salmonella in cheese from El Salvador, shrimp from Costa Rica, and frozen lobster tails from Honduras; and mercury in fresh swordfish from Costa Rica. There is no indication of continued, large-scale detentions of agricultural products from Central America due to pesticide residues, except for snowpeas from Guatemala as mentioned above.

Table 6 shows how the U.S. determined its general import sampling strategy for 1996. These figures roughly correspond to relative importance of the country's import values in the U.S. As a result, Guatemala and Costa Rica are sampled relatively frequently compared to Honduras, El Salvador, and Nicaragua.

Clearly, given past experience with detentions and growing concerns in the U.S. about the safety of its food system, there are additional costs on exporters and importers due to added inspection and handling requirements. These costs of business exist now, and will remain in the future. Even in the absence of the past detention experience, it seems highly likely that these additional costs would exist anyway because much of the concern of pesticides in food in the U.S. is driven by the quality of food produced in the U.S., not just imported foreign food.

for pesticides among the Central American countries included in this study, the country's detention value for 1993 is not necessarily representative.

[30] For reference, during the same month, there were 171 FDA detentions for product from Taiwan, 166 from Taiwan, 114 from the United Kingdom, and 141 from Germany, 103 from South Korea, and 347 from China. These are detentions of all types, such as inadequate packaging or labeling for the U.S market, not just pesticide concerns.

III. ON-FARM COSTS OF SOIL EROSION/DEGRADATION

Soil erosion is a natural process that occurs without human land use.[31] It is commonly understood that soil erosion from agricultural fields can reduce soil productivity (i.e. cause soil degradation), which in turn reduces crop yields. As Lutz, Pagiola, and Reiche (1994, editors' preface) point out: "it is often suggested that soil erosion and degradation are among the most severe environmental problems in developing countries…."

On-farm productivity effects are the most important consequence of soil erosion. For example, Pagiola and Dixon (1997, p. 1) summarize conventional wisdom: "land degradation is thought to be the most important natural resource management problem in El Salvador...Land degradation is thought to lead to reduced agricultural productivity and sedimentation of reservoirs." While concerns about on-site productivity effects of soil erosion have existed for some time, Lutz, Pagiola, and Reiche (1994, p. 273), however, conclude that "land degradation is thought to pose a severe threat to the sustainability of agricultural production. Yet despite long-standing concern about this threat and dramatic claims of environmental damage, surprisingly little empirical analysis has been done on the causes and severity of land degradation problems in the region (Central America and the Caribbean) and on how best to tackle them".[32]

It is not clear, however, if on-site productivity impacts of soil erosion and degradation should be considered an externality. Consider the case where soil erosion or degradation has only on-site effects. If agricultural land values reflect the present value of future agricultural incomes earned from the land, which is basically the case when there are secure property rights, land markets, and good information, then any future impacts on productivity are incorporated into existing production decisions. As a result, while there may be some level of soil erosion or degradation on existing lands over time, there are no additional costs associated with excessive levels of erosion and degradation.

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[31] Crosson (1983, p. 17) points out that the Missouri River in the U.S. was known as the "Big Muddy" at a time when its drainage basin was occupied by "scattered Indian tribes, a handful of mountain men, and a few isolated outposts of the United States Army".

[32] Reiche (1996), which uses existing information on total soil degradation based on the work of Oldeman, et al. (1990), suggests that the leading causes of soil degradation in Central and South America are: conversion of native forests to agriculture and livestock (34% of 'degraded' area); forestry activities (7.5%), grazing (25%) and agricultural production activities (30%, presumably on existing agricultural land as opposed to converted lands included in the first category).

This simple model is not valid for many reasons, including that land markets in rural areas are not always well functioning, property rights are not well enforced, and information needed to understand the on-site effects of soil erosion and degradation can be scarce or difficult for farmers to evaluate.[33] The extent to which these problems, however, lead to too much erosion and degradation is not clear.

At a fundamental level, the impacts of soil erosion and degradation on agricultural productivity are difficult at best to document for several reasons. First, clear definitions of and/or data on erosion and degradation do not exist. Consequently, the relationships between degradation and productivity are not clear either.

For example, consider erosion. The concept is clear--the number of tons of soil removed from a field in a given time period. Measuring erosion, however, is not a simple task. Until recently, most estimates of soil erosion were based on either the Universal Soil Loss Equation (USLE) or estimates based on sediment delivered to watershed outlets. The USLE was developed from test plot data in the U.S. Midwest. Recent evidence, however, suggests that it may not estimate soil loss well in different situations.

Estimates based on sediment delivered require a delivery ratio for the watershed. Since such ratios may vary between 45% of eroded soil in small watersheds to only 5% in large watersheds, the estimated total erosion (or average erosion per hectare) can vary substantially. Moreover, even if the delivery ratio is known, it does not provide information on from where the soil originated. For example, Crosson (1983) cites that in the Los Laureles watershed in Honduras, an estimated 2% of land area in roads and trails contributed 45% of all soil delivered to a water outlet.

Even if soil erosion is well estimated for agricultural fields, the link between erosion and soil fertility decline (degradation) is not clear. At a fundamental level, notions of soil fertility, quality, and degradation are ill-formed. For example, a university textbook called <u>Soil Fertility and Fertilizers</u> (Tisdale, Nelson, and Beaton, 1985) does not even offer a definition of soil fertility.[34] It does suggest that the most important 'environmental' factors know to influence

[33] Crosson (1983) points out that sheet and rill erosion is a gradual process that is not easily detected. He notes that an erosion rate of 50 tons per hectare, which is large relative to rates found in often-reported field tests from El Salvador in the late 1970s (Pagiola and Dixon, 1997), reduces soil depth by about 0.363 centimeters per year. Erosion rates of 100 tons per year would still represent less than a one centimeter reduction is soil depth.

[34] See, for example the definitions in Carls, Reiche, and Jauregui (1997) taken from the Soil Science Society of America.

plant growth include: temperature, moisture supply, radiant energy, composition of the atmosphere, soil structure and composition of soil air, soil reaction, biotic factor, supply of mineral nutrient elements, and absence of growth-restricting substances.

While these factors are accepted as important, the ability to develop more complete models of plant growth and nutrients has been limited (Tisdale, Nelson, and Beaton, 1985, p. 53). A simple continuous relationship is not adequate. Instead, the "law of the minimum" remains a powerful concept for understanding plant growth and, as a result, productivity. For example, soil acidity or limited moisture will limit yield response to additional fertilizer. At the same time, certain substitution possibilities could overcome these limits. For example, Tisdale, Nelson, and Beaton (1985) report that corn yields on subsoil depended to a large degree on the permeability of the subsoil, not simply that surface soils were eroded away. With adequate permeability and added lime, phosphorus, and potassium, corn yields were 95% of those grown on top soils.

The links between soil erosion, soil degradation, and yields on control plots are not simple and are not well understood. As a result, it is not surprising that good, generalizable information on soil erosion, degradation, and productivity decline in actual situations is not available, although the literature does contain some scattered information.

Conceptually, a correct definition of the on-site economic cost of soil degradation at one point in time would involve the following experiment. First, the level of inputs and crop choices that maximize profits, for example, per hectare for a given level of soil quality must be evaluated. Call this initial level of per-hectare profits B(Q', M') where Q' is the initial level of soil quality and M' represents input and output choices that maximize net benefits B. The cost of soil degradation when Q falls from Q' to Q'' is just B(Q', M') - B(Q", M"), where input and output levels also adjust as quality changes. Using some elementary microeconomic relationships, this term just equals the value of the change in yield associated with a very small decline in yields. For larger changes it is necessary to include input and output adjustments as well. The estimates of large yield changes holding all other inputs constant, denoted as B(Q', M') - B(Q", M'), is an overestimate of the costs of soil erosion.[35] Thus, the basic recommended methods for estimating the soil erosion-productivity relationships, for example as discussed in

[35] In a more dynamic context, if soil quality declines now affect yields now and in the future, the cost of soil degradation now would essentially be equal to the present value of current and future profit reductions.

Olson, Lai, and Norton (1994) which varies soil characteristics but holds everything else constant, is not enough for economic analysis of the costs of soil erosion.

Some efforts have attempted to estimate on-site economic costs of soil erosion in developing countries, working within the substantial gaps in understanding between soil erosion, degradation, and resulting productivity changes. For example, Magrath and Arens (1989) estimate such costs for the island of Java in Indonesia. The baseline range of estimates for on-site productivity effects in relation to soil erosion makes the Java report quite relevant to Central American agriculture. Lacking clear information on actual erosion rates (tons per hectare), the Java case had to estimate erosion based on soil types, cropping practices, and location in the country. Average erosion rates were predicted as 123 tons per hectare for agricultural fields across the country that were dry land farming on sloping fields. The erosion ranged from 76-144 tons per hectare. Given these erosion estimates and informed guesses concerning the relationship between yield and soil erosion, the report estimated that on average there would be between a 4-7% yield decline (approximately) due to soil erosion at these rates.

Magrath and Arens (1989) note that actual yields during 1972-1983 increased for main crops 2.8-4.7 percent per year on average in part due to increase fertilizer use and higher yield varieties. This point emphasizes a basic issue: actual yield trends are not necessarily correlated with productivity impacts of soil erosion. Thus, increasing yields do not imply that erosion has had no impact on productivity. Similarly, falling yields do not imply strong effects of soil erosion.

Based on their erosion estimates and informed guesses, the 4-7% decline in yields over time due to estimated soil erosion levels are valued at 4% of agricultural net income on average across Java. (The estimate varies from 1% in central Java to 10% in West Java.) It is likely that these estimates of the cost of erosion in West Java could be considered an upper bound on the general importance of soil erosion costs in terms of on-farm productivity in Central America across the agricultural sector as a whole.

These cost estimates reflect total on-farm productivity costs of erosion. They do not in themselves provide any direct information on how much soil erosion should be reduced. These numbers do, however, provide useful information on the benefits of reducing soil erosion. Whether it is economically useful to reduce erosion depends on the costs of erosion reduction.

The apparent dearth of information on actual field level estimates of soil erosion in Central America is evident in a review paper by Lutz, Pagiola, and Reiche (1994). Their Table 1 provides some empirical evidence on estimated soil erosion rates and reflects the paucity of available information. This study's Table 1 reports only eight site-specific studies completed during 1956-1991 in Central America (three in Panama, one in Nicaragua, two in El Salvador, and two in Honduras). In hindsight, Pagiola and Dixon (1997) show that the information reported as "average" in Lutz, Pagiola, and Reiche (1994) for one study in El Salvador was actually the maximum observed on one plot for one year and not representative of the study. In the end, Lutz, Pagiola, and Reiche (1994) conclude that good data on soil degradation and productivity effects is scarce.

While good data are few, there have been attempts at evaluating the on-site costs of soil erosion in Central America. Solorzano et al. (1991) estimated an average erosion rate of 70 tons per hectare over the 1970-89 period and that soil erosion 'cost' about 7% of agricultural GDP (roughly 1% agricultural GDP per 10 tons of erosion). While these estimates are useful, the approach used in Solorzano et al. (1991) does not actually estimate the cost of soil erosion. Instead, Solorzano et al. (1991) estimate the loss of main soil nutrients associated with estimated erosion levels, and then calculate the costs of replacing these nutrients with inorganic fertilizers. The cost of replacing soil nutrients, which is the cost of one strategy to increase yields, is not the same as the cost of soil erosion. Such replacement costs are probably higher than on-farm erosion costs.[36]

Cardona and Zuñiga (no date) develop a mathematical model relating soil depth to yields of broccoli. While it is not clear how the data related to soil depth and yields were developed (test plots or just 'created'), the 14 observations included information on soil depth ranging from 3.5 to 15 inches, with seven plots each on 'high' and 'low' soil fertility. For these 14 points, broccoli yields ranged from about 3.6 to 16.3 MT/Ha. With the 14 observations, a simple OLS regression can be estimated as $y = 0.39 + 0.55x + 5.9D$, where y is yield, x is soil depth in inches, and D is a dummy equal to zero on "low" fertility plots and equal to one on "high" fertility plots.[37]

[36] Recall also that the 'law of the minimum' in plant growth could imply that replacing some nutrients would have a yield impact if they are the limiting factor, but replacing non-limiting nutrients would have no yield impact.

[37] For reference, the t-statistics for the two estimated parameters were about 6.8 and 9 respectively, the r-squared was 0.92, with 14 observations and 11 degrees of freedom.

This simple estimation shows the clear importance of having fertile soil as distinct from soil depth. Yields on fertile but shallow soils were almost the same as yields on deep but low fertility soils. Since x is in inches, the estimate suggests that a one-inch reduction is soil depth (e.g. due to soil erosion) would reduce yield by 0.55 tons per hectare. If 100 tons of erosion implies about a 2/7inch reduction in soil depth (Crosson, 1983), then average soil loss of about 34 tons (as reported as average for broccoli in this study) would have a yield impact of about 100 kg of broccoli per year (assuming 34 tons is net soil loss). This would equal about 1% of yield on the more productive plots and almost 3% of yield on low fertility plots (holding all other inputs constant). If farmers had no other means of improving soil fertility, then yields of broccoli would be estimated to fall quickly overtime. Fortunately, soil fertility and depth can be restored over time through crop rotations, fertilizers, etc. Again, it should be emphasized that the data in Cardona and Zuñiga are 'synthetic' data. It is not clear how they reflect reality.

Considering recent information on soil erosion and soil conservation in El Salvador , Pagiola and Dixon (1997) suggest that 75% of the country's surface is degraded. Pagiola and Dixon (1997) note that such conclusions are based on almost no empirical data (e.g., an estimate for one year on one plot generalized to a large scale), and that there is no obvious aggregate yield decline for maize. As noted elsewhere, such as in Magrath and Arens (1989), yield trends bear no direct relation to the actual cost of soil erosion.

The data summarized in Pagiola and Dixon (1997) are also important because they show that farmers have fairly clear opinions on the relationship between soil erosion and yield changes over time. Whether their opinions are correct or not, of course, is not at all clear. Nonetheless, it does indicate that farmers have some important knowledge of the relationship between erosion and productivity, and that they do undertake soil conservation practices that are privately profitable and ignore others that are not (Lutz, Pagiola, and Reiche, 1994). As a result, it is very likely that on-site productivity effects are included in farmer decision making.

Based on data on the sample of farmer opinions, Pagiola and Dixon (1997, p. 13) conclude that perhaps one-third of fields on moderate slopes (accounting for 30% of surveyed fields) and two-thirds of fields on steep slopes (accounting or 10% of surveyed fields) experience "productivity problems." Data are not available to venture a guess about the level of productivity impacts, but they note that it seems that farmers have been able to manage such effects through adjusting other inputs. Perhaps most important in this study is that 60% of fields surveyed are not on moderate or steep slope-sand presumably therefor have with fewer erosion problems.

Most of the work on on-site productivity costs of soil erosion is carried out in a simple partial equilibrium framework. A more recent study by Alfsen et al. (1996), however, suggests that the overall costs of soil degradation, in terms of effects on GDP growth at the national level, could be larger than indicated by the partial equilibrium calculations. The study also suggests that the costs could be distributed across the economy outside of the direct ‘agricultural’ sector. The study, however, can just be considered suggestive at this point because all the information used to estimate soil erosion levels and yield declines are based essentially on a informed guess of one person and the production structure of the model is fairly restrictive.[38]

With these caveats in mind, Alfsen et al. (1996) assume that annual productivity declines for major crops in six regions of Nicaragua are about 2% per year for beans, 1-2% for maize and sorghum, 1% for coffee, less than 1% for sugar and tobacco and vegetables, and 1-3% for pasture. Based on these estimated annual yield declines (defined as a ‘hicks neutral’ shift down in the agricultural production function), the results in Alfsen et al. (1996) suggests that annual GDP growth over the period 1991-2000 would fall from 2.9 % to 2 %. As a result, estimated real GDP in 2000 would be about 7% lower than in the baseline scenario.

Alfsen et al. (1996) also note that some economic groups, particularly land owning peasants who are net sellers of agricultural products actually benefit from soil erosion because the output declines are translated into even higher output prices because of inelastic consumer food demands. Since the majority of the population is urban, the majority of costs of soil degradation are borne by landless laborers (rural and urban) and urban sectors. This model indicates that the links from soil erosion to economic growth could be strong over time in an agricultural dependent economy. More developed computable general equilibrium (CGE) frameworks and better data on erosion and productivity impacts are still necessary to evaluate such impacts.

[38]The production structure in the model is also based on a constant-returns to scale Cobb-Douglas production function.

IV. CONCLUSIONS

The main objective of this paper was to examine existing empirical evidence on the extent of environmental and health impacts of agricultural production in Central America. The focus here is on impacts that occur at the intensive margin, in large part because it can be expected that most agricultural growth will come from these lands in the future.[39] Within this general topic, the on-site productivity effects of soil erosion and the public health effects of pesticide use are the two main types of external costs in the literature on agricultural production in Central America.

Soil Erosion

Regarding the negative effects of soil erosion on agricultural productivity, even though such concerns have existed for many years, the links between erosion, soil attributes, soil degradation, and productivity remain poorly understood. To date, the literature shows that the on-site productivity effects of soil erosion are not substantial for existing agricultural lands and that farmer's make appropriate soil conservation efforts as rational investment decisions.[40] There are clearly examples of severe erosion in the region, but there are no indications that such concerns are an important constraint now and in the future for agricultural growth opportunities.[41]

Health Effects

Regarding the health effects of pesticide use in Central America, the literature remains incomplete, politically sensitive in some places, and difficult to generalize to a national or regional level (e.g. Central America). Based on the review and additional analysis presented in this paper, however, a reasonable guess values the direct costs of just acute pesticide poisonings

[39] As mentioned in the text of the paper, off-site and extensive margin externalities are clearly important and are addressed in a separate paper.

[40] Of course soil erosion and lost productivity on more newly converted agricultural lands (on hillsides, tropical forested areas) have been substantial. While traditional methods of shifting cultivation were specifically adapted to such problems, newer attempts at essentially sedentary production (crops and pastures) on such lands in Central America shows that methods are not available or at least profitable where the majority such lands can be continuously used over time. In these cases, soil erosion is probably better viewed as another externality from land conversion/deforestation rather than a by-product of agricultural production per se.

[41] Again, the distinction needs to be clear between the newly converted lands in marginal areas as opposed to the more-or-less long term agricultural lands. While this distinction may not be black and white, it is clear that some lands are currently in crops that cannot be sustainably managed at a reasonable high level of productivity.

in Central America at 2.5% of total agricultural GDP (about $190 million in 1995). From this total cost, almost 50% is due to suicide-related intentional deaths, 25% is due to accidental deaths, and about 25% is due to non-fatal poisonings. (See Table 3 for details.) Thus, at least based on the information in the existing literature, which is focused mainly on acute health effects, the costs of pesticide-related deaths rank substantially above the costs of non-fatal poisonings.[42]

Information on the chronic effects of acute poisonings and chronic effects of longer-term low dose exposures is not adequate to hazard a guess at economic costs at this time. There is clear awareness that some substantial portions of the population in Central America are exposed to and receive doses of toxic chemicals that persist in the environment over long time periods (e.g. organochlorines such as DDT). The literature reviewed on ambient monitoring studies (surface waters, ground waters, aquatic plants and animals, and people) shows that pesticide levels in the ambient environment can vary widely over time and space. Household-level panel data are clearly needed to begin to develop a better understanding of the impacts of chronic impacts of pesticides, and the responses of rural households to such chronic effects.[43]

Next Steps and Policy Initiatives Regarding Health Impacts

As mentioned in the introduction to this paper, it is necessary in principle to evaluate both the benefits and costs of any actions or policies designed to reduce agricultural externalities (e.g. acute health risks from pesticide poisonings). While such economic efficiency logic is clear, as a practical matter it can help substantially to have a better appreciation of the magnitude of the external costs before considering options to reduce them for at least two reasons. First, it is often not feasible to evaluate many different types of external costs and many options for reducing each type of cost before taking action. As a result, it usually makes sense to begin to rank

[42] The difficulties associated with understanding and evaluating pesticide-related health effects cannot be underestimated. First, there are a vast number of pesticides available on the world market. Second, basic scientific understanding of potential human health effects from various pesticides remains partial before and after pesticides are sold on markets. Third, even if dose-response relationships are known, the health effects of pesticide poisonings can be very similar to other medical causes. As a result, under reporting is clearly an issue that will continue to exist. And fourth, there has been almost no analysis of the correct economic costs associated with pesticide-related health risks, which in principle would involve a direct understanding of willingness to pay to avoid such health risks

[43]Even though information on chronic effects sparse, the substantial use of highly toxic chemicals with little use of protective gear suggests that more analysis is needed of worker risk attitudes, avoidance activities, and willingness to adopt various risk reduction methods.

potential problems (in terms of estimated magnitude of external costs), and then begin to focus first on the higher priority problems.

Thus, in terms of this review and additional analysis, it is clear that acute pesticide poisonings are an important cost to the economies of Central America, and specifically deaths from suicides and then accidental deaths. If the costs associated with reducing the risks from such deaths are considered to be reasonable, it seems safe to suggest that carrying through with such actions is appropriate. The next step for future work is to evaluate specifically the costs of reducing risks of pesticide related deaths (suicides and accidental poisonings as separately). To address such topics, it will be necessary for agricultural policy makers to work with other parts of government to develop and implement effective responses. Issues of non-lethal accidental poisonings are a second level issue, in part because the costs associated with such effects are not estimated to be large at this time and because individuals have some ability to take averting actions to avoid some of these risks.[44]

It is also clear that Central America continues to be exposed to pesticides that are persistent in the environment but are currently banned in the region and elsewhere. The existing literature provides examples where the residues of such chemicals are found at very high levels. Given the continued concern over such issues, the lack of consensus of their importance and absence of any appropriate public policy response, a valuable first policy step is the commission a regional-level risk assessment. This screening-type population health risk assessment should be based on currently available risk assessment techniques and information, and should focus on a short list of banned persistent pesticides, e.g. DDT. Such an assessment should also focus on at least two pathways (water and food consumption), although others could be included if warranted. It is also clear that some substantial levels of extrapolation and possible scenarios would be needed to complete such an assessment. Once a clearer picture of the risks, externalities and costs is available, options for reducing such risks can be identified, evaluated, and the costs and be compared to other public health priorities.

[44] Besides taking actions to avoid such risks, there is also some ability to mitigate/reduce the health impacts following the poisoning event.

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APPENDIX I. A REVIEW OF EXTERNALITIES

That private and social costs can differ extends back to at least Adam Smith in the *Wealth of Nations* but was developed more specifically by Pigou and Marshall (see, e.g., Bromley, 1986). A main reason for a divergence between private and social costs--the externality-- is that the actions of some person, firm, or group imposes real physical effects on some other person, firm, or group. The fact that these physical effects are imposed on others (i.e. external to the person making the decision) implies that distributional issues are often central to policy discussions associated with such physical effects.

Of course physical effects do not always translate directly into some monetary notion of cost or benefit. For example, soil erosion provides an easy example of the difference between physical effects and some monetary cost. Consider two plots of land in different locations with different soil depths experiencing the same quantity of erosion per year. On plot A the topsoil depth is high, the farming cycle replaces needed materials adequately with few costs, and the soil erodes onto forested areas lower in the watershed. As a result, the observed physical effect (soil erosion) involves essentially no costs (on farm or off farm). Plot B is just the opposite. Plot B has shallow soils and the farming cycle does not replace needed materials adequately. Moreover, the plot is located near a reservoir used for hydroelectric production. As a result, there are clear costs associated with erosion on plot B including perhaps reduced profits in the future from the agricultural productivity effects of soil erosion as well as impacts on costs of electricity production. Thus, for the same physical effect, say 250 tons of erosion annually, the external cost on plot A is zero and the cost on plot B is not. This example is intended to emphasize that physical effect is not the same as its associated monetary cost.

As noted above, the concept of externality begins with the idea that private and social costs can differ. Consider a simple example of pesticide use outlined in Figure 1. In Figure 1, pesticide use is denoted as x kilograms of active ingredient of some toxic compound (e.g. aldrin, chlordane, dieldrin, etc.). Farmers use pesticides because they perceive an economic benefit associated with pesticides as well as direct costs of purchase and application (including any safety measures). Assuming that farmers try to maximize *their* benefits minus *their* costs, the efficient level of pesticide use occurs at a point where their marginal private benefits (denoted as MB) equal their marginal private costs (denoted as MPC). This outcome, denoted as x' in Figure

1, is referred to as the 'efficient private outcome'. At the efficient private outcome, where MB = MPC, x' is the efficient level of pesticide use (or soil erosion, etc.). At x', the total private net benefits (private benefits minus private costs) to agriculture equals the area C+D+E.

Figure 1 also includes a marginal total social cost curve (MSC) which is above the marginal private cost curve (MPC) due some additional external costs of pesticide use. In Figure 1, the difference between MSC and MPC is defined a marginal external costs (MEC). Taking a simple hypothetical situation without pesticide regulations, these marginal external costs are associated with pesticide use that farmers *can* ignore because they do not pay the costs (the cost exists, just that farmers do not pay the costs). In Figure 1, at the level x' of pesticide use, the additional external costs of pesticides use at x' equal C+D+F. As a result, net total social benefits at pesticide use x' are E-F. These total social benefits can be called the 'total economic value' of pesticide use (at level x').

Because external costs are ignored by farmers in Figure 1, the level of pesticide use x' is too high. When marginal benefits are equated with marginal total costs MSC in Figure 1, the socially efficient level of pesticide use is x*. At this level of pesticide use, private benefits to farmers are E+C costs to farmers are A, external costs paid by others are C, and net social benefits are E. Note that net social benefits at x* are greater than net social benefits at x' by the term F. The information in Figure 1 is also summarized in Table 1.[45]

For reference, four different types of external costs can be considered in Figure 1: (1) external costs at the privately efficient outcome (the area C+D+F); (2) external costs at the socially efficient outcome (the area C); (3) external costs created by using pesticides at a level beyond the socially efficient level (D+F); and (4) the reduction in net social benefits of using pesticides at level x' instead of x* (the area F).[46] It is important to be clear on these four types of costs when analyzing agricultural externalities and when reviewing the existing literature. For example, while total external costs at existing levels of pesticide use (the term C+D+F) may be large in Central America, the reduction in social costs from 'internalizing the externality' and reducing use from x' to x* (just F) could be substantially smaller. Also, for example, while the external costs C at the socially efficient use x* are real, there is no simple economic efficiency justification to reduce pesticide use beyond x*.

[45]While Figure 1 is developed based on the idea of external costs, it is also possible to consider the case of external benefits, in which private benefits do not adequately reflect social benefits.

[46] The term F equals the cost to the economy of not 'internalizing' the externality.

As an introduction to the literature, the following conclusions can be made:

(1) For most environmental and health issues related to agriculture, there is not adequate information on the existing situation in physical terms (denoted as x' in Figure 1) and empirical analysis to date is based on sporadic case studies and educated guesses. Unfortunately, while better information on these physical effects is needed, such information will remain relatively imprecise in the foreseeable future. As with other policy problems, the lack of hard data now does not necessarily justify a 'wait-and-see' approach to dealing with the problems.

(2) For all agricultural externality topics included in this paper, there is little understanding of, or agreement on, socially acceptable levels (denoted as x* in Figure 1), in large part because the MSC function and in many cases the MB function in Figure 1 are not well understood. While not directly included in the analysis in this paper, there is widespread concern that physical effects (either x' or x* in Figure 1) are too large due to government policy, for example through tax breaks and other subsidies. This makes the marginal private benefit function too high and/or marginal private cost function too low in Figure 1.

(3) Relying on inadequate data, there are some estimates of existing levels x'(e.g. quantity of pesticide use, quantity of acute pesticide poisonings, tons of soil erosion) but essentially no useful estimates of the value of external costs associated with the privately efficient level (denoted as C+D+F in Figure 1). Such cost calculations are needed to evaluate the relative economic returns of various sectors of the economy or different subcomponents of agriculture.

(4) There are effectively no estimates of the efficient outcomes x*, the external costs associated with such an outcome x* (denoted as B+B' in Figure 1), and the external costs associated with production beyond the socially efficient level (denoted as D+F).

(5) There are no estimates of the net social costs of ignoring external effects and operating at the level x' rather than the social level x* (the term F in Figure 1).

APPENDIX II. TABLES AND FIGURES

Table 1. Potential Externalities from Agricultural Use of Pesticides

Acute Health Impacts (Morbidity and Mortality)	Chronic Health Impacts
Intentional Poisonings (Suicides)	Morbidity following from poisoning High dose
Accidental, non-work poisonings	Morbidity following Low-Dose over time
Occupational Poisonings	Mortality Following Low-Dose over time

Direct Market Costs to Economy	Indirect Market Costs to Economy
Pesticide residues on Agricultural Exports	Reduced labor productivity following pesticide poisoning
Pesticide residues in animal and fish products	Reduced cognitive ability following poisoning with future impacts on productivity
Fish kills from Pesticides in Surface water	Increased pest resistance to pesticides, with resulting increases in pest control costs both for agricultural and public health protection

Table 2. General Information on Central American Countries

Variable	Units	Costa Rica	El Salvador	Guatemala	Honduras
Population	*(millions)*	3.4	5.6	10.6	5.9
1995 GNP	*per capita ($)*	2610	1610	1340	600
1995 GNP	*per capita ($ PPP)*	5850	2610	3340	1900
Country Rank in WDR 97	*Number*	81	70	63	40
Labor force	*(millions)*	1	2	4	2
% Labor Force in Agriculture		0.26	0.36	0.52	0.4
GDP 1995	*(billions $)*	9.2	9.4	14.4	3.9
% Ag GDP		0.17	0.14	0.25	0.21
% Ag Annual GDP Growth Rate	*(1990-95)*	3.6	1.2	2.5	2.9

Source: World Bank, WDR, 1997.

Table 3. Implications of Annual Acute Pesticide Poisonings in Central America

3.A MORBIDITY (Annual)	World Rate	Costa Rica Rate	High End Rate
Annual Poisoning Rate (ag workers)	0.03	0.045	0.08
Agricultural Labor Force	4420000	4420000	4400000
Annual Poisonings	132600	198900	352000
Central America Malaria Cases (annual)	139338	139338	139338
Economic Cost per Poisoning	200	200	200
Lost Wages			
Medical Services			
Pain and Suffering			
Total Cost: All Acute Poisonings	**$26,520,000**	**$39,780,000**	**$70,400,000**
Percent Intentional Poisonings	0.24	0.24	0.24
Total Cost: Non-intentional Poisonings	**$20,155,200**	**$30,232,800**	**$53,504,000**
Total GDP in Region (billions $)	38.8	38.8	38.8
Total Ag. GDP in Region (billions $)	7.926	7.926	7.926
Cost as Percentage of Ag. GDP (1995)	**0.25**	**0.38**	**0.68**

3.B MORTALITY (Annual)	Pesticides Exc. Suicides	Pesticides Total
Costa Rica Annual Mortality Acute Poisonings	60	60
Costa Rica Percent from Suicides	0.62	
Non-suicides	22.8	
Costa Rica Ag Labor Force	260000	260000
Percentage Mortality Rate	0.000088	0.0002
Total Central America Ag Labor Force	4420000	4420000
Total Mortality--Pesticides	388	1020
Occupational Deaths Total		1790
Traffic Deaths		3648
U.S. Value of Statistical Life ($ Millions)--Example	3	3
Ratio of Central America to U.S. per-capita GNP	0.05	0.05
Central America VOSL ($)--Guess	150000	150000
Total Cost Based on VOSL	**$58,140,000**	**$153,000,000**
Total GDP in Region (billions)	38.8	38.8
Total Ag. GDP in Region (billions)	7.926	7.926
Cost as Percentage of Ag. GDP (1995)	**0.73**	**1.93**

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Table 4. Pesticides Detectable and Found (*) in 1996 FDA Residue Monitoring Report[a,b]

Acephate*
Acetochlor
Acrinathrin
Alachlor
Aldicarb*
Aldrin
Allethrin
Allidochlor
Alpha-cypermethrin
Ametryn
Aminocarb
Amitraz
Anilazine
Aramite
Atrazine
Azinphos-ethyl
Azinphos-methyl*
Bendiocarb
Benfluralin
Benodanil
Benomyl/carbendazim[c]
Benoxacor
Bensulide
Benzoylprop-ethyl
6-Benzyladenine
BHC*
Bifenox
Bifenthrin*
Binapacryl
S-Bioallethrin
Biphenyl*
Bitertanol*
Bromacil

Bromophos
Bromophos-ethyl
Bromopropylate
Bromoxynil
Bufencarb
Bulan
Bupirimate
Butachlor
Butralin
Butylate
Cadusafos
Captafol*
Captan*
Carbaryl*
Carbofuran*
Carbophenothion
Carbosulfan
Carboxin
Chlorbenside
Chlorbromuron
Chlorbufam
Chlordane*
Chlordecone
Chlordimeform*
Chlorethoxyfos
Chlorfenapyr
Chlorfenvinphos
Chlorflurecol methyl ester
Chlorimuron ethyl ester
Chlornitrofen
Chlorobenzilate
3-Chloro-5-methyl-4-nitro-1H-pyrazole
Chloroneb

Chloropicrin*
Chloropropylate
Chlorothalonil*
Chloroxuron
Chlorpropham*
Chlorpyrifos*
Chlorpyrifos-methyl*
Chlorthiophos
Clomazone
Coumaphos
Crotoxyphos
Crufomate
Cyanazine
Cyanofenphos
Cyanophos
Cycloate
Cycluron
Cyfluthrin
Cymoxanil
Cypermethrin*
Cyprazine
Cyproconazole
DCPA*
DDT*
Deltamethrin
Deltamethrin, trans
Demeton*
Desmetryn
Dialifor
Di-allate
N,N-Diallyl-dichloroacetamide
Diazinon*
Dichlobenil

Dichlofenthion
Dichlofluanid
Dichlone
4-(Dichloroacetyl)-1-oxa-4-azapiro[4.5]decane
2,6-Dichlorobenzamide
Dichlorvos*
Diclobutrazol
Diclofop-methyl
Dicloran*
Dicofol*
Dicrotophos
Dieldrin*
Diethatyl-ethyl
Dilan
Dimethachlor
Dimethametryn
Dimethipin
Dimethoate*
Dinitramine
Dinobuton
Dinocap
Dioxabenzofos
Dioxacarb
Dioxathion
Diphenamid
Diphenylamine*
Dipropetryn
Disulfoton
Diuron
Edifenphos
Endosulfan*

Endrin*
EPN*
Esfenvalerate*
Etaconazole
Ethalfluralin
Ethephon
Ethiofencarb
Ethion*
Ethofumesate
Ethoprop
Ethoxyquin*
Ethylenebisdithiocarbamates*[d]
Etridiazole
Etrimfos
Famphur
Fenamiphos
Fenarimol
Fenbuconazole
Fenfuram
Fenitrothion*
Fenoxaprop ethyl ester
Fenoxycarb
Fenpropathrin*
Fenpropimorph
Fenson
Fensulfothion
Fenthion
Fenvalerate*
Fipronil
Flamprop-M-isopropyl
Flamprop-methyl
Fluazifop butyl ester
Fluchloralin
Flucythrinate
Flusilazole
Fluvalinate
Folpet*
Fonofos*
Formothion
Fosthiazate
Fuberidazole
Furilazole
Gardona
Heptachlor*
Heptenophos
Hexachlorobenzene*
Hexaconazole*
Hexazinone
Hexythiazox
Imazalil*
Imazamethabenz methyl ester
Iprobenfos
Iprodione*
Iprodione metabolite isomer*
Isazofos
Isocarbamid
Isofenphos
Isoprocarb
Isopropalin
Isoprothiolane
Isoxaben
Isoxaflutole
Lactofen
Lambda-cyhalothrin
Lenacil
Leptophos
Lindane*
Linuron*
Malathion*
Mecarbam*
Mephosfolan
Merphos
Metalaxyl*
Metaldehyde*
Metasystox thiol
Metazachlor
Methabenzthiazuron
Methamidophos*
Methidathion*
Methiocarb*
Methomyl*
Methoprotryne
Methoxychlor*
2-Methoxy-5,6-trichloropyridine
Metobromuron
Metolachlor
Metolcarb
Metribuzin
Mevinphos*
Mirex
Monocrotophos*
Monolinuron
Monuron
Myclobutanil*
Naled*
Napropamide
Neburon
Nitralin
Nitrapyrin
Nitrofen
Nitrofluorfen
Nitrothal-isopropyl
Norea
Norflurazon
Nuarimol
Octhilinone
Ofurace
Omethoate*
Ovex
Oxadiazon
Oxadixyl
Oxamyl*
Oxydemeton-methyl
Oxyfluorfen
Oxythioquinox
Paclobutrazol
Paraquat*
Parathion*
Parathion-methyl*
Pebulate
Penconazole
Pendimethalin
Pentachlorobenzene*
Pentachlorobenzonitrile
Pentachlorophenyl methyl ether*
Permethrin*
Perthane
Phenothrin
Phenthoate*
Phenylphenol, ortho-*
Phorate*
Phosalone*

Phosmet*
Phosphamidon*
Phoxim oxygen analog
Piperonyl butoxide*
Piperophos
Pirimicarb
Pirimiphos-ethyl
Pirimiphos-methyl*
Pretilachlor
Probenazole
Prochloraz
Procyazine
Procymidone*
Prodiamine
Profenofos*
Profluralin
Prolan
Promecarb
Prometryn
Pronamide
Propachlor
Propanil
Propargite*
Propazine
Propetamphos
Propham
Propiconazole
Propoxur
Prothiofos
Prothoate
Pyracarbolid
Pyrazon
Pyrazophos*
Pyrethrins
Pyridaphenthion
Pyrimethanil
Quinalphos*
Quintozene*
Quizalofop ethyl ester
Ronnel
Schradan
Secbumeton
Simazine*
Simetryn
Strobane
Sulfallate
Sulfotep
Sulphenone
Sulprofos
TCMTB
Tebuconazole
Tebupirimfos
Tecnazene
TEPP
Terbacil
Terbufos
Terbumeton
Terbuthylazine
Terbutryn
Tetradifon
Tetraiodoethylene
Tetrasul
Thiabendazole*
Thiazopyr
Thiodicarb
Thiometon
Thionazin
Thiophanate-methyl
THPI*
Tolylfluanid
Toxaphene
Tralomethrin
Traloxydim
Triadimefon*
Triadimenol*
Tri-allate
Triazamate
Triazophos
Tribufos*
Trichlorfon
Tricyclazole
Tridiphane
Trietazine
Triflumizole
Trifluralin*
Triflusulfuron methyl ester
Trimethacarb
Vamidothion sulfone
Vernolate
Vinclozolin*
XMC

[a] The list of pesticides detectable is expressed in terms of the parent pesticide. However, monitoring coverage and findings may have included metabolites, impurities, and alteration products.
[b] Some of these pesticides are no longer manufactured or registered for use in the United States.
[c] The analytical methodology determines carbendazim, which may result from use of benomyl or carbendazim.
[d] Such as maneb.
Source: Food and Drug Administration Pesticide Program, Residue Monitoring Report 1996, U.S. FDA, Center for Food Safety and Applied Nutrition Pesticide Program Monitoring Report 1996, January, 1998, retrievable at http://vm.cfsan.fda.gov.

Table 5. Frequency of Occurrence of Pesticide Residues Found in Total Diet Study Foods in 1996[a]

Pesticide[b]	Total No. of Findings	Occurrence, %
DDT	140	18
Malathion	136	17
Chlorpyrifos-methyl	122	16
Endosulfan	87	11
Dieldrin	76	10
Chlorpyrifos	72	9
Chlorpropham	45	6
Iprodione	36	5
Carbaryl[c]	33	4
Methamidophos	32	4
Dicloran	27	3
Thiabendazole[d]	27	3
Permethrin	23	3.0
Dimethoate	22	2.8
Acephate	21	2.7
Dicofol	21	2.7
Lindane	21	2.7
Diazinon	19	2.4
BHC	18	2.3
Toxaphene	18	2.3

[a] Based on 3 market baskets analyzed in 1996 consisting of 778 items. Only those found in >2% of the samples are shown.
[b] Isomers, metabolites, and related compounds are not listed separately; they are covered under the "parent" pesticide from which they arise.
[c] Reflects overall incidence; however, only 93-95 selected foods per market basket (i.e., 283 items total) were analyzed for N- methylcarbamates.

[d] Reflects overall incidence; however, only 65-67 selected foods per market basket (i.e., 199 items total) were analyzed for thiabendazole and benomyl.

Table 6. Foreign Countries and Number of Samples[a] Collected and Analyzed in 1996

Mexico	1752	**Taiwan (Formosa)**	44
Chile	409	**Philippines**	42
Canada	329	**Unspecified**	37
Netherlands (Holland)	238	**Japan**	36
Guatemala	186	**France**	35
Thailand	175	**Argentina**	30
China, Peoples Rep.	172	**Australia**	29
Italy	164	**Brazil**	27
Costa Rica	141	**Jamaica**	27
India	123	**Hong Kong**	24
Spain (inc. Canary Islands)	123	**Germany, Federal Rep.**	22
Dominican Republic	107	**United Kingdom**	21
Ecuador	91	**Lebanon**	20
Peru	79	**South Africa**	20
Israel	70	**Poland**	18
Colombia	67	**Venezuela**	18
Panama	67	**Morocco**	17
Turkey	59	**El Salvador**	15
Indonesia	51	**Pakistan**	15
Greece	49	**Nicaragua**	13
Korea, Rep. Of (South Korea)	49	**Denmark**	12
Honduras	47	**Trinidad & Tobago**	11
Belgium	44	**Viet-Nam, Rep. Of**	11
New Zealand	44		

Figure 1. Private and Social Efficiency