Part III. Assessing IPM Impacts

Introduction

The second theme of the Third National Symposium/Workshop, "Assessing IPM Program Impacts," was motivated by several factors. First, the Clinton Administration's commitment to implementing IPM practices on 75 percent of crop acres by the year 2000 has put a spotlight on defining and measuring the degree and extent of IPM adoption in the United States. Second, the concomitant goal of reducing reliance on high-risk pesticides to garner environmental and public-health benefits demands new methods of measuring pesticide impacts. Third, to meet the demand for greater accountability for public expenditures (as legislated in the Government Performance and Results Act passed by Congress in 1993), the USDA IPM Initiative and National IPM Implementation Plan require integration of assessment activities in future IPM funding proposals.

Careful documentation of IPM program impacts can help demonstrate that recommended IPM technologies and practices are both profitable for producers and reduce reliance on agricultural chemicals that are harmful to the environment and/or public health. While the need for better documentation of IPM program impacts is clear, a consensus has not yet been forged about the appropriate assessment method(s) to use. Past efforts to evaluate IPM-program impacts have generally focused on the cost and efficacy of IPM practices. Environmental impacts were often limited to measuring pesticide-use reduction. Enlarging the assessment domain to include broader concepts of environmental and public-health impacts adds additional complexity that can best be addressed by the adoption of multidisciplinary assessment approaches.

USDA officials and a private consultant presented their views on integrating multidisciplinary assessment into IPM research and extension programs in the plenary session, "Assessing IPM Program Impacts." These opening comments were followed by the presentation of five papers commissioned by the Economic Research Service that provided a starting point for an interdisciplinary discussion of the appropriate methods and approaches for measuring economic, environmental, and public-health impacts of IPM programs. Each of these papers is published in its entirety here. ERS also organized five selected paper sessions during the Symposium/Workshop that provided a venue for the presentation of empirical and methodological research results exploring some aspect of IPM evaluation. A summary of each paper presented is provided at the end of this section of the Proceedings.

Karl Stauber, (former) Under Secretary for Research, Education, and Economics, and Susan Offutt, Administrator of the Economic Research Service (ERS), opened the plenary session devoted to assessment. In their introductory comments, the Under Secretary and the Administrator expressed the Department of Agriculture's commitment to supporting multidisciplinary assessment of IPM impacts. Stauber, in his program overview, "Interdisciplinary Collaboration to Achieve IPM Goals," highlighted the importance of establishing IPM research and extension priorities that reflect both producer needs and public concern about agriculture's effects on environmental quality and human health. He argued that accountability for the use of public funds will require a transparent assessment process that documents progress toward achieving priorities identified by all the stakeholders. In his view, IPM adoption offers producers and society a potential win-win solution by maintaining producer profits and addressing environmental and public-health issues associated with pesticide use.

Terry Nipp, President of AESOP Enterprises, Ltd., in his presentation, "Accountability: The Best Defense is a Good Offense," underscored the importance of establishing an open assessment process that documents progress toward the achievement of societal priorities. In his view, agricultural research and extension programs that can demonstrate benefits to producers and improvements in meeting important societal goals (such as environmental protection, worker safety, safe water and food, and wildlife protection) will have a higher probability of retaining and maybe even increasing their public funding.

Having made the case for integrated interdisciplinary assessment, the authors of the five commissioned papers addressed alternative assessment approaches. Integrating different disciplinary perspectives into a coordinated assessment was the challenge undertaken in the paper by John Antle and Susan Capalbo, "Integrated Assessment of IPM Impacts: An Overview." Because no single technology will be superior in all assessment areas, a unifying framework is needed to assess the tradeoffs among economic, environmental, and public-health impacts of alternative production technologies. The authors described how physical impacts, once identified, can then be converted into monetary values, thereby providing a common unit of measurement. They then explain how to use a benefit-cost framework to assess the tradeoffs between different objectives.

Susan Riha, Lois Levitan, and John Hutson in "Environmental Impact Assessment: The Quest for a Holistic Picture" outlined the issues that must be addressed in assessing pesticides' impacts on the environment. They discussed objectives, strengths, and weaknesses of existing environmental assessment methods and identified conceptual and data challenges that must be overcome to improve these assessment tools. Important issues (such as who is going to use the assessment, time frame, budget, and the tradeoffs between ease-of-use versus complexity and short run versus longrun) were identified as important questions useful in determining the appropriateness of alternative ap-proaches and tools in environmental assessment.

The many challenges encountered in trying to measure and assess acute and chronic health impacts of occupational exposure to pesticides are explored in "Occupational Exposure to Pesticides and Their Effects on Human Health" presented by Aaron Blair, Marcie Francis, and Sarah Lynch. The authors reviewed current public-health research on the relationship between occupational exposure to pesticides and the development of acute and chronic diseases, including cancer and diseases of the nervous, immune, and reproductive systems. Understanding how and to what degree pesticide exposure occurs [source(s), route(s), duration, and dose] is critical to estimating public-health impacts.

A more detailed description of how to conduct economic-impact assessment is provided in "A Primer on Economic Assessment of Integrated Pest Management" by George Norton, Jeffrey Mullen, and Edwin Rajotte. The authors walk readers through the "nuts and bolts" of conducting an integrated economic assessment, including a process for defining IPM systems, identifying appropriate assessment methods, establishing statistically valid baseline data, and integrating and analyzing this information in a benefit-cost framework. While recognizing that the site-specific nature of IPM systems means that a standardized approach to measuring impacts is not possible, the authors identified a core set of methods that can form part of virtually any IPM impact assessment. They also presented an overview of some of the methods that are available to address other dimensions of an integrated assessment.

Farm-level profitability and technical efficiency are two powerful factors influencing producer adoption of new technologies. However, IPM practitioners have been puzzled by the lack of adoption of some IPM practices or technologies that have been both profitable and efficient. In "Practical Considerations in Assessing Barriers to IPM Adoption," Peter Nowak, Steven Padgitt, and Thomas Hoban identified other considerations besides economic and technical efficiency that influence adoption of alternative agricultural practices. The authors argued that IPM is an information-intensive production system. Deepening and expanding the use of IPM will depend on increasing the number of producers who *want to* and *can* incorporate site-specific, multifaceted information in their pest-management decision making. Viewing IPM as a decision-making process rather than as a list of practices makes the task of measuring adoption vastly more complex. The authors presented a typology of barriers to adoption of IPM practices that differentiates between producers who are unable, unwilling, or both unwilling and unable to adopt IPM systems. If gains are to be made in deepening and expanding adoption of IPM, then understanding the important differences between the reasons for not adopting recommended IPM practices will contribute greatly to the identification of appropriate policies and strategies.

While each of the presentations and commissioned papers dealt with different aspects of impact assessment, collectively they identified key elements that must be addressed in conducting integrated assessment. First, because of the diversity in agroecosystems, IPM systems, weather, and pest pressures, appropriate methods may need to be adapted to reflect site-specific conditions. Second, because of this diversity, each locale must develop a consensus on assessment priorities through an open, transparent process that includes all stake-holders. Budget constraints and data availability limit what can be studied, so agreement must be reached by stakeholders on what is to be assessed and how. Third, relevant disciplines must be included at the start of the research project to allow researchers to agree upon a common unit of analysis for data collection, scientifically valid data-collection procedures, spatial and temporal scales, and complementary methods to quantify the impacts of IPM production technologies. Working together from the start will facilitate the integration of the different disciplines' methodological approaches into a comprehensive assessment. Fourth, converting impacts into a common monetary measure facilitates the comparison of different impacts and the assessment of tradeoffs between different objectives. Finally, an assessment must quantify the economic, environmental, and public-health impacts of IPM adoption and show the regional and socioeconomic distribution of these impacts.

Interdisciplinary Collaboration to Achieve IPM Goals

Karl Stauber
Former Under Secretary, USDA

I would like to add to that of the Administrator of the Economic Research Service, Dr. Susan Offutt, my welcome to participants of the Third National IPM Symposium/Workshop. An important theme of this conference is "Meeting the IPM Goal." The conference program reflects the importance of two important elements identified by the USDA IPM Initiative as critical to the success of meeting this administration's IPM goals. The first, "Putting Customers First," means that priorities for IPM research and educational programs must reflect our customer-identified needs. These needs must be identified through a systematic planning process involving all stakeholders. The second, "Incorporating Impact Assessment," implies that the successful implementation of the IPM Initiative will require us to carefully document the environmen- tal, economic, public-health, and social impacts of increased IPM implementation by farmers and other IPM users.

The USDA IPM Initiative is a coordinated Department-wide effort to realize the Clinton administration's goal of implementing IPM practices on 75 percent of the nation's crop acres by the year 2000. This goal, set jointly by the Department of Agriculture, the Environmental Protection Agency, and the Food and Drug Administration in the fall of 1993, reflects the administration's commitment to improving environmental quality while maintaining the agricultural sector's profitability and global competitiveness. The administration has backed this commitment with increased budget proposals to support IPM research and extension education programs in both FY 96 and FY 97 budget requests. The proposed increases are the first significant increases for IPM research and extension activities since the Nixon administration. The USDA Strategic Plan for the IPM Initiative commits the Department to provide research, educational, and programmatic support to address priority needs identified by farmers and other IPM stakeholders.

The 75-percent IPM goal has stimulated a great deal of discussion as to its origin and what it means in terms of a measurable goal. This goal must be viewed in the context of public concern about environmental quality, food safety, and the use of pesticides by both agricultural and urban users. Several European countries have mandated pesticide-use-reduction goals in response to similar concerns about pesticide impacts on the environment and public health. The Administration's 75-percent goal depends on voluntary adoption of IPM practices rather than mandated use-reduction goals. It emphasizes the proven track record of the land-grant-university system as an agent of innovation and change. In addition, the administration's goal focuses on the potential for IPM to reduce farmer reliance on pesticides while enhancing economic and environmental benefits to producers and society as a whole.

The IPM Initiative, carried out by the research, education, and economics mission area of USDA, will provide increased support for basic and implementation research and educational programs needed to encourage voluntary adoption of IPM systems. The IPM Initiative will not only reach out to new adopters of IPM practices but will provide support for present IPM users to incorporate more sophisticated IPM tactics on their farms.

The 1994 Economic Research Service report on IPM adoption indicates that basic IPM tactics are used on approximately 50 percent of U.S. crop acres. This might indicate that we are two-thirds of the way to our goal. I prefer a more ambitious interpretation. While many American farmers have adopted some basic IPM tactics, we need to invest in focused research and education programs to provide the foundation for new farmers to adopt IPM production practices and at the same time provide existing IPM users with a range of more comprehensive IPM tactics to adopt.

A new report from the National Research Council encourages the adoption of "ecologically based IPM." To promote the adoption of ecologically based IPM we must commit ourselves to a significant public investment in both research and extension education. It is clear that achievement of ecologically based IPM or the simpler goal of implementation of IPM on 75 percent of the crop acreage will require integrated program planning that involves both the biological and social sciences if the IPM Initiative is to be responsive to the complex demands placed on agriculture in today's society. This Initiative epitomizes the type of approach that will be increasingly demanded by the public to address a variety of issues in the agricultural sector. Why? Because pest-management issues are elements of a broad array of multidimensional challenges that agriculture confronts: protection of natural resources and the environment, viability of rural communities, sustainability, public investment in agricultural research, education and farm programs, and global competitiveness. The USDA, in cooperation with its land-grant-university partners and a broadly defined user community, must create a coordinated strategy to engage both disciplinary science and interdisciplinary system-oriented approaches to address increasingly complex agricultural problems.

Public concerns over agriculture's effects on environmental quality and human health must be addressed in planning and implementing the IPM Initiative. Also important, however, is the need for producers to achieve sustainable economic returns for their investment. By involving all of IPM's stakeholders in a dialogue, we can address the private-risk, public-benefit paradigm. The adoption of IPM practices can provide a win-win solution to pest problems by maintaining producers' economic viability and global competitiveness and at the same time addressing environmental and public health issues associated with pesticide use.

The key to expanded IPM adoption is to understand that IPM practices and technologies are site-specific and both knowledge- and information-intensive and that producers will not adopt unprofitable practices. The IPM Initiative will succeed if it focuses its resources on research and education priorities identified at the local level by producers and other stakeholders. Critical to the success of the Initiative is the establishment of an assessment process that documents progress toward achieving the priorities identified by the stakeholders. Information derived from the assessment process improves accountability and contributes to a better understanding of the factors that contribute to both success and failure.

I have asked the Economic Research Service, working through the USDA IPM Coordinator and IPM Program Subcommittee and with other USDA agencies and the EPA, to take the lead in formulating an assessment plan for the IPM Initiative. This plan will help with assessment at both the national and local level and will require the unique disciplinary expertise of both the biological and social sciences and the forging of new interdisciplinary alliances.

This conference offers an opportunity to increase our understanding of the components of successful IPM programs and the environmental, economic, public-health, and social impacts of IPM programs. The dialog and planning initiated during this symposium/workshop will contribute both to strengthening disciplinary science and forging the synergistic new interdisciplinary alliances needed to achieve the administration's IPM goal. I will watch with interest how the challenges of "Putting Our Customers First" and "Incorporating Impact Assessment" are addressed in the IPM plans being developed at both the state and production-region levels. I and other members of the administration will work with Congress to bring the needed new resources for research and education to your local programs.

Integrated Assessment of IPM Impacts: An Overview

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Introduction

The purpose of this paper is to provide an overview of how the economic, environmental, and public-health benefits and impacts of IPM can be measured and used in an integrated assessment of IPM. Before addressing how this can be done, it is important to explain why it should be done, particularly because most IPM researchers do not consider impact assessment a part of IPM research, and it has not been included in most IPM research projects.

There are a number of important reasons why we need to do integrated assessment of IPM impacts (see Antle and Wagenet 1995 for a more detailed discussion). First, from the scientific perspective, we need information on the expected benefits and costs of alternative research strategies to set research priorities, to design research, and to evaluate research. In short, to do good science, we need to use resources efficiently; and to do that, we need to be able to assess how productive science is. There is also a need for this information to conduct policy research.

Second, there is a growing demand by the public and by government for publicly funded research, such as IPM research, to be socially and economically accountable. Executive orders under the Reagan, Bush, and Clinton administrations have required accountability for major new regulations and policies, and Congress has required similar accountability under the Government Performance Review Act. The need for this information is particularly acute to justify expenditures on publicly funded research, such as IPM, in an era of declining government spending on research, and it is needed to set priorities among competing research programs. Indeed, USDA's IPM Initiative is built on the premise that development and adoption of IPM will yield economic, environmental, and human-health benefits to producers and to society. Obviously, it is USDA's responsibility to demonstrate that the research sponsored by this program actually achieves those objectives if this line of research is to justify continued funding.

Researchers naturally tend to view impact assessment as a burdensome, costly task that diverts resources from scientific work. But this view of impact assessment is mistaken on several grounds. First and foremost, this view is much like the person who is in such a hurry to get somewhere that he does not bother to look at the map. How can we defend the claims made for the benefits of IPM if we do not document them? Second, if there really are substantial economic, environmental, and public health benefits from IPM, IPM researchers have a strong vested interest in having those benefits quantified and documented. It would be myopic, indeed, for IPM researchers not to view impact assessment as an essential part of the IPM research agenda. Finally, there is a tendency to view economics, environmental science, and health science as not part of IPM and therefore as detracting from the pool of money available for IPM research. This view ignores the fact that in a world where publicly funded science must be justified by the benefits it yields, there may be no pool of money for any kind of IPM research if the benefits cannot be documented and quantified in a scientifically sound manner.

In the remainder of this paper, we address the question of how to do integrated impact assessment for IPM research. There are two essential points that we would like to emphasize in our discussion of impact assessment:

Impact assessment must be an integral part of doing IPM research and extension and must be integrated into research and extension projects from their inception:

- to facilitate interdisciplinary collaboration in the design and implementation of data collection and analysis;

- to ensure that the research is useful and relevant in economic, environmental, and public-health terms;

- to ensure that the impact assessments are timely and cost-effective. It is often argued that impact assessment is too time consuming and costly. This is not true if impact-assessment research is integrated into research projects from their inception.

Impact assessment is an application of the economic tool of benefit-cost analysis, combined with appropriate data and models from production economics, environmental science, and health science.

Because it is difficult to value all of the environmental and health impacts, impact assessment should strive to quantify tradeoffs among economic, environment, and health impacts. These tradeoff relationships can be used to assess the benefits associated with IPM technologies.

The Impact-Assessment Framework

Benefit-cost analysis provides the basis for a multidisciplinary approach to assessing impacts of IPM and other research activities (Antle and Wagenet 1995). Note that the use of "multidisciplinary" is meant to convey the need for collaboration across the full spectrum of biological, physical, and social sciences that are needed to address the impacts of agricultural technology. The first step is for scientists to set research objectives that reflect public priorities. We shall describe these objectives broadly as food supply, human health, and environmental. The public's priorities may be embodied in state or federal legislation or may be communicated to research administrators and scientists by local interest groups, such as commodity, farm, or environmental organizations. Researchers then formulate strategies to meet these objectives. For each strategy, researchers collaborate to estimate the impacts of the prospective technologies on production, human health, and the environment.

Once impacts are estimated by each discipline, economists can translate the impacts into monetary values. The valuation of market goods, like wheat, is straightforward because market prices can be used. The monetary valuation of nonmarket goods, such as environmental amenities, is more difficult but can be done in some cases and is a major component of environmental-economics research (Freeman 1993). The present and future benefits and costs of the prospective technologies are translated into present values with a technique known as discounting. This technique weights monetary benefits and costs by a discount factor that takes into consideration how far into the future the benefit or cost occurs. These discounted benefits and costs are then summed over time. The difference between discounted benefits and costs of each strategy is its net present value (NPV).

Because agricultural research is an uncertain undertaking, the ultimate value of research to society is also uncertain. Researchers must consider the probability of success of each research strategy and uncertainties associated with estimating benefits and costs of research. For example, taking into account the scientific and economic uncertainties, each research strategy may be associated with a pessimistic (low) NPV value and an optimistic (high) NPV value. Weighting these possible NPVs by their probability of occurrence yields the statistical expected value of the NPV. Research strategies are ordered according to their expected NPV, and only projects with a positive expected value are considered acceptable. When some of the impacts, such as changes in human health or the environment, defy quantification or valuation in monetary terms, a qualitative assessment can supplement the quantitative analysis.

A number of issues that cannot be treated here in detail must be considered in implementing impact assessment. One critical issue is identifying the distribution of benefits and costs across the affected groups. For example, the economics literature considers how research conducted in one geographic region affects productivity in other regions. An important part of environmental and health impact assessment is identifying the relevant population. Another issue arises when public research is an input into the private development of technology. In this case, the research contributions from both the public and private sectors must be determined.

Assessing Impacts of Pest-Management Research

To illustrate how impact assessment can be designed and used in IPM research, let us now consider the challenge of designing pest-management research to accomplish the sustainable agriculture goals in the 1990 farm bill. As we noted in the Introduction, one important motivation for impact assessment is the need to set research priorities. We consider two research strategies. One is based on genetic manipulation of the plant to resist a pest, such as the development of late-blight-resistant potato varieties, which if successful would eliminate the need for certain classes of pesticides, such as the fungicides used to control late blight; the other is based on a conventional IPM strategy, such as improving the timing and amount of fungicides applied to potato crops, that may reduce but does not eliminate pesticide use.

A successful pest-management strategy must be profitable to individual farmers and for the indus-try as a whole if it is to be widely adopted. In collaboration with the biological researchers, economists can estimate changes in pesticide use, labor, other inputs, and yields associated with the two research strategies. The extent of adoption of the technology by the industry and its economic impact at the farm and industry level can then be estimated. Many such studies of IPM have been conducted by agricultural economists (e.g., Carlson and Wetzstein 1993).

The human-health and environmental impacts of a change in pest-management technology also can be quantified. Despite the public perception that IPM techniques reduce or eliminate pesticide use, many IPM techniques are based on "economic thresholds" for pesticide application that do not explicitly consider either environmental or human-health impacts. The agricultural-science community tends to assume that environmental and health problems associated with technologies are caused by inefficient use of the technology. Inefficient use may indeed be one source of health and environmental problems, as in pesticide use by farmers in developing countries. But even the correct use of an "economic threshold" could result in overuse of a pesticide when off-farm environmental or health effects are considered. These "external costs" are particularly important in policy design because they are not borne by farmers and the market does not provide an economic incentive for farmers to take corrective actions.

Teams of economists, occupational-health specialists, and environmental scientists can assemble data on human toxicity of the pesticides, their transport, and fate in the environment. These data can be used to estimate changes in human-health risk, water quality, and other key dimensions of health and the environment associated with the IPM technologies and the use of recombinantly derived resistant varieties. If the agricultural products are traded internationally, international standards for pesticide residues and the use of genetically altered materials must be considered in the estimation of benefits and costs. If the data on the economic, health, and environmental benefits are combined, the net present value (NPV) of each technology can be estimated.

Various outcomes are possible in this example, depending on the weights attached to crop production, environmental quality, and health. If both strategies yield a positive expected NPV and if the research budgets are adequate, then both strategies might be funded to account for the uncertainties in research. If the biogenetic research strategy is more costly and the benefits of reduced pesticide use are not large or if its success is highly uncertain, then the less-costly, more-reliable IPM strategy might be preferred. But if the health or environmental costs of using pesticides are sufficiently large, the benefits of the biogenetic strategy that could eliminate the use of pesticides might yield the higher expected NPV. It is also possible that neither line of research could yield sufficiently high benefits to justify its cost.

Designing Integrated Assessments: Units of Measurement and Aggregation

It should be apparent from the preceding discus-sion that researchers involved in an interdisciplin-ary project must coordinate their research designs so that data can be integrated across disciplines and used for impact assessment. We assume that the production impacts of prospective technologies have been quantified by agricultural scientists. Soil and crop science tell us that the environmental benefits of reduced pesticide use vary according to soil and climatic conditions. The pesticide-reducing technologies will be adopted by many farms operating in widely differing climatic conditions and soils. Thus, pesticide impacts vary across the physical and economic units in production. Likewise, public health researchers know that the human health impacts of pesticides vary across individuals in the affected populations. How can we quantify the benefits of technologies whose impacts vary across space or time?

This question raises a fundamental issue in the design of research for impact assessment. Biological and physical science research typically focus on the cellular, plant, animal, or field level. This level is different than the level at which technologies affect the public and at which public policies are directed. Even policies at the local level will be directed at a population of biological, physical, or economic units. In water policy, for example, federal law requires states to assess impacts and to formulate policies at the level of a well-defined environmental entity, such as a watershed or aquifer.

The solution to this problem is for researchers from all concerned disciplines to be involved at the inception of the research, so that they can agree upon a unit of analysis to use in quantifying the impacts of production technologies. In the water-quality example, soil scientists, and economists can define a unit of measurement, such as a farmer's field, at which both the economic and environmental impacts of the technologies can be reliably assessed. The physical impacts in the population of farm fields can be described by probability distributions of solute leaching below the root zone and runoff into surface water. Economists can also estimate in probabilistic terms how farmers change pesticide use as they adopt the new pesticide-reducing technologies. By combining these physical and economic data for the physical and economic populations, it is possible to estimate the mean environmental impacts in the population or to assess the probability that leaching or runoff will exceed a critical level. This environmental-risk information can then be related directly to policy objectives.

Assessing Impacts:
The Role of Tradeoffs

Identifying the impacts of production technologies on human health and the environment takes us a significant step closer to making the link from science to impact assessment and policy formation. But in both research planning and impact assessment, it is rare that one research strategy or technology dominates all others in all relevant dimensions. One technology may be more productive but also riskier for human health than another; thus, tradeoffs among economic, health, and environmental goals must be assessed.

One solution to this problem is to obtain a common unit of measurement by converting physical impacts to monetary values. The use of monetary values is appealing because the economic impacts of a technology on producers and consumers--changes in net returns to producers and changes in the real incomes of consumers--can be measured with market prices. Government policies often distort market prices, so analysts must consider these distortions.

Health and environmental impacts of technology create an additional valuation problem. The monetary valuation of changes in human health and environmental quality usually cannot be measured directly because these are nonmarket goods. The valuation of nonmarket goods has been a major research objective in environmental economics for the past 30 years. An established set of techniques now exists to obtain values for nonmarket impacts that are comparable to market prices.

There are, however, several significant limitations to the application of nonmarket valuation techniques. First, the transferability of valuations is an unresolved issue in the economics literature, and it may be prohibitively costly to undertake a valuation study corresponding to every nonmarket effect that needs to be considered in an impact assessment (Larson 1995). Second, the reliability of the valuation techniques has been questioned in the economics profession, and the economic valuation of some nonmarket effects is controversial in the public mind and may not be accepted by the public as a basis for impact assessment (Smith 1992; Portney 1994).

For these reasons, we believe it is important for researchers conducting impact assessments to present tradeoffs among economic, environmental, and public-health impacts whether or not nonmarket valuation techniques are used to translate impacts into monetary terms.

How It Is Done: Assessing the Economic, Environmental, and Health Tradeoffs of Pesticide Use in Potato Production

We now illustrate the impact assessment methods outlined above by describing a study designed to assess the economic, environmental, and health effects of pesticide use in potato production. Detailed descriptions of this study can be found in Antle, Crissman, and Capalbo (1994); Crissman, Cole, and Carpio (1994); and Antle et al. (1996).

This study of the economic, environmental, and human-health effects of pesticides sponsored by the International Potato Center was based in the Carchi Province in northern Ecuador in a highland zone 30 km south of the Colombian border. Production occurs between the altitudes of 2,800 and 3,400 m on steeply sloped, deep volcanic soils. Just half a degree north of the equator, there are virtually no changes in day length, little seasonal variation in temperature, and limited variation in rainfall.

The cropping system is dominated by potatoes and pasture for dairy cattle, with these two crops rotated in a potato-potato-pasture cycle that takes about 2 years. Because of the equatorial Andean climate, there are no distinct planting or harvesting seasons, and potato production occurs continuously. Production data were collected in a farm-level survey on 40 farms during 1990 to 1992 by trained enumerators who lived in the region and made bimonthly visits to the farms. Data were collected for individual parcels, where a parcel is defined as a single crop cycle on a farmer's field.

This physical environment is highly conducive to certain potato pests, notably the soil-dwelling larvae of the Andean weevil (*Premnotrypes vorax*) and the late-blight fungus (*Phytophthora infestans*). With backpack sprayers, farmers make an average of more than seven applications of pesticides to each parcel. Though a wide array of products was used, three types dominated the selection. The dithiocarbamate Mancozeb accounted for more than 80 percent of total active ingredient of fungicides. The carbamate Carbofuran and the organophosphate Methamdiophos accounted for 47 percent and 43 percent of all insecticide active ingredients applied. Carbofuran is used to control the Andean weevil, and the organophosphates are used on foliar insect pests. Most farmers manage several fields, so that potato production and pesticide use are continuous throughout the year. An important consequence of continuous production is a year-round potential for occupational and incidental exposure to pesticides. Pesticides are not used in the pasture cycle and are seldom used in other crops that may be included in the rotation, such as legumes. Thus a farmer's exposure to pesticides comes almost entirely from potato production.

The project's research team consisted of agricultural economists, soil scientists, and occupational health researchers. In the planning stage of the project, the study watersheds were identified, and the decision was made to collect production data at the field level. Detailed parcel-level production data were collected on a monthly basis, with emphasis on accurate measurement of pesticide use. An important part of the production work was to account for the fact that a large number of different types of pesticides are used in the production system. The watersheds were classified into four agroecological zones, and soils, and related data were collected by the environmental impact team for simulation modeling of the transport and fate of pesticides in the environment.

To examine the health impacts of this pesticide use, the health research team conducted a survey of the farm population and an age- and education-matched reference group not exposed to pesticides. All participants answered questions on pesticide use and medical problems, received a clinical examination by a field physician, completed a series of tests of nervous system function, and underwent blood tests. These tests were oriented toward those effects most likely to be associated with the insecticide and fungicide exposures that the agricultural team had documented. Crissman, Cole, and Carpio (1994) describe the higher rates of skin problems (dermatitis), reduced vibration sensation, lower cholinesterase levels, and generally poorer neurobehavioral test results among the farm population compared to the reference group.

Following the approach described by Antle, Capalbo, and Crissman (1994), primary production data were used to estimate econometric models that represent the farmers' decisions on the extensive (crop choice) and intensive (input use) margins. These econometric models provided the parameters for construction of a stochastic simulation model of the production system. The outcomes of this economic simulation model were then input into two other simulation models: a physical simulation model to estimate environmental impact, defined here in terms of the leaching of pesticides beyond the crop root zone; and a simulation model based on statistically estimated relationships between pesticide use on the farm and the neurobehavioral status of members of the farm population.

These three integrated simulation models were used to assess the economic, environmental, and farm-population health impacts of various scenarios, including alternative pest-management scenarios. Simulation-model output can be displayed in graphs that illustrate the tradeoffs between agricultural output and changes in environmental quality (e.g., leaching of an insecticide below the root zone) for the current management practices and an IPM practice that involved more effective carbofuran application techniques. Similarly, the tradeoffs between agricultural output and health risk under current management practices, under the IPM technology, and for a combination of IPM and improved farmworker protection practices can be constructed. In this particular study, these tradeoff relationships showed that there are substantial tradeoffs among output, environmental, and health outcomes and that IPM practices improve health as much as or more than better self-protection practices. In other words, this case study showed that IPM could generate substantial benefits by reducing numbers of insecticide applications and thus lowering exposure to hazardous insecticides.

Conclusions

In this paper, we argue that impact assessment must be an integral part of doing IPM or any other publicly funded agricultural research. Impact assessment does not take resources away from IPM research, rather it is an integral part of doing research that addresses society's concerns about the impacts of agriculture on environmental quality and public health. A key goal of impact-assessment research should be to quantify tradeoffs among economic, environmental, and public-health outcomes.

Another important message we would like to convey to the research community is that we must not be overwhelmed by the apparent complexity of these problems. Successful research programs will use experts from each relevant discipline to identify key first-order impacts in each area (economic, environment, and health) and focus on them. Interdisciplinary collaboration at the research design stage will also ensure that units of measurement are compatible across disciplines so that research results can be integrated for impact assessment.

Finally, it must be emphasized that in impact assessment, as in all scientific research, there is no cookbook solution. The general approach described here must be adapted to each production system to account for its most important impacts.

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Environmental-Impact Assessment: The Quest for a Holistic Picture

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Agriculture intentionally disturbs the natural ecosystem and imposes a managed system that has multiple direct and indirect environmental consequences. Given the uncertainty and complexity of these consequences, a number of different approaches for assessing the impacts of agricultural practices on the environment have been proposed and discussed. All these methods can be viewed as attempts to answer the question "What are the environmental consequences of agricultural management decisions?" IPM investigators are currently being challenged to respond to this question as part of their research and as one means of assessing the success of IPM. Previously, IPM has been judged primarily in terms of the cost and efficacy of IPM practices. To the extent that environmental impact was considered, it was assessed primarily by reduction in pesticide use or by indicators important to implementing IPM (for example, the impacts on beneficial arthropods).

The objectives of this paper are twofold: first, to encourage IPM investigators to think more deeply about the potentials, limitations, and complexities of environmental-impact assessment and, second, to acquaint IPM investigators with the range of current approaches they might use to evaluate environmental impacts of their IPM programs. The paper is divided into four sections. The first section discusses the meaning of environmental impact. Our purpose is to inspire researchers to think broadly when considering environmental impacts, and to illustrate some of the consequences of a narrow view of the environment. The second section describes a number of challenges in conducting environmental-impact assessments. The point of this section is to encourage researchers to recognize problems with current environmental assessment methods and to use these as a motivation for improving assessment tools. The third section presents a typology of approaches to environmental assessment. We discuss the objectives, strengths, and limitations of various assessment methods but do not evaluate particular environmental assessment methods. This section is meant to encourage researchers to consider how different types of assessment methods may or may not be suitable for their project. The last section considers some practical issues that researchers face in deciding which assessment method to use. These issues include determining who the assessment is supposed to serve and trade-offs in ease-of-use versus complexity. The aim of this section is to encourage researchers to consider these issues explicitly before choosing an environmental assessment method.

Defining Environmental Impacts

When we refer to environmental impact, what comes to mind will differ depending on one's view of the environment and the components of the environment that one values. Environmental-impact assessments measure or estimate impacts on one or more environmental indicators. Many groups are concerned with assessing the degree to which various components of the environment are changing. However, different groups may have a particular interest in particular components of the environment and little interest in others. We have chosen to review several concepts that we hope will encourage researchers to think more broadly when considering what is meant by the environment and which environmental variables might be assessed for impact. These concepts include (1) how newer ideas differ from the classic ecotoxicological model, (2) how we focus on events that occur in various places in space and time, and (3) the physical resource base. Environmental impacts can be thought of as including all nontarget impacts; but for the purposes of this paper (and following the EPA Science Advisory Board, see Cooper 1993), we are not considering human-health effects as environmental impacts.

Chemical to Biocriteria

When considering if a pesticide application has had an environmental impact, we might first think in terms of how the application of pesticides on the farm affects the pesticide concentration in ground and surface waters, the atmosphere, and soils. Pesticide input on the farm can be related to pesticide concentration in the environment by applying a fate model that predicts how the pesticide will move from where it is applied to the environment of interest. The concentration of the pesticide in the environment is then related to potential impact on specific biota with toxicity ratings and some type of exposure factor. Traditionally, ecotoxicology has focused on single-species toxicity testing in the laboratory to develop repeatable thresholds of response to changes in toxin concentration and exposure (Cairns 1995). These tests have the advantage of linking a biological response to a specified level of toxin and, therefore, in theory, can maintain a link between a farm-management decision (e.g., pesticide application) and a biological response (e.g., death of fish). The impact on biota established through such tests (e.g., an LD_{50}) are referred to as test endpoints (Suter 1995). If chemical concentration exceeds a toxicity threshold for one or more species, then the environment is considered to be impacted. This approach to defining environmental impact is summarized in figure 1a.

One of the reasons that the classic ecotoxicological model has been widely used is that it is easier to set goals and write regulations related to chemical levels (e.g., in terms of the concentration of pesticide in groundwater) than in terms of impacts on ecosystems. Objections have been raised to the individual-species toxicity tests that are integral to this model. These objections include: the limited array of species used may not be most sensitive, the same species is not most sensitive to all chemicals, and species may respond differently when not isolated from other species (Cairns 1995). Micro- and mesoscale testing systems have been developed to overcome some of these objections. The results of these tests have been considered by some too inconsistent to be practicable, although Cairns (1995) believes this approach may have been too easily dismissed. More generally, the classic ecotoxicological model fails when the acceptable level of a chemical in the environment as established from test endpoints does not correlate with the environmental impacts of interest to the public. Another shortcoming of applying the classic ecotoxicology model to assessments of agricultural impact is that people are generally not directly concerned with the level of a chemical in the environment per se, even if this level is lethal to 50 percent of a specific organism in a test. What is of interest to them is the impact of management decisions on such components of the environment as populations of biota and the functioning of ecosystems (Karr 1995), which are sometimes referred to as assessment endpoints (Suter 1995). We will use the term decision endpoints in referring to these environmental components that are of actual interest to various decision-making groups.

In response to some of the limitations of the classic ecotoxicological model, with its focus on chemical criteria, some scientists suggest using field-measured biological criteria that can be more directly related to decision endpoints (Karr 1995) rather than single-species toxicity tests (Fig. 1b). The use of biological criteria as indicators of environmental impact has both a public and a scientific tradition. For centuries, people have been concerned about fish supplies and more recently have expressed concern for the preservation of other wildlife (Policansky 1993). There is increasing public and scientific interest in the more general notion of environmental integrity and a recognition by the scientific community that single-species toxicity is not necessarily indicative of system-level responses (Policansky 1993; Barbour et al. 1995; Cairns 1995). Characterizing environmental integrity generally requires measures of an array of biological attributes. These can include use of habitat indices, conditions of individual organisms (i.e., diseases, anomalies, or metabolic processes), community structure measures (i.e., taxa richness and trophic dynamics), and productivity measures. In environmental assessment, this approach has probably been taken furthest in evaluating the integrity of water resources (Barbour et al. 1995).

Although biocriteria are important indicators of environmental impact, their use raises several problems. There is not currently a widely accepted, multidimensional measure of biological integrity/ecosystem quality (Barbour et al. 1995). An index of biotic integrity (IBI) has been developed with biosurvey data to construct a multimetric index of heterogeneous variables (Karr 1981; Simon and Lyons 1995). Criticisms of this index approach include ambiguity, eclipsing of one metric by another, arbitrary variance, unreality involved in combining unlike metrics, post hoc justification, single linear scale of response, inability to use in diagnostics, and nonsense results. Simon and Lyons (1995) attempt to defend IBI in the face of these criticisms, but many of Suter's concerns are inherent to such indices and therefore should be taken seriously.

A second problem in the use of biocriteria is in defining appropriate reference conditions, particularly in terrestrial ecosystems (Policansky 1993; Hughes 1995). The problems encountered in defining reference conditions can be easily illustrated by issues in restoration ecology. To what condition should derelict or degraded land be restored? Both in restoration ecology and in defining an acceptable biological status of an ecosystem, it has been recognized that human values must be taken into consideration. Diamond (1987), in his studies of restoration ecology, points out that different segments of the population hold different values and therefore different views of appropriate restoration conditions. Hughes (1995) position is that "The [biological] reference condition must be politically palatable and reasonable. In other words, it must be acceptable and understandable by persons most concerned with nature for its own sake and those unconcerned with nature or only concerned with what it can provide humans. If the process for determining the reference condition is acceptable and understandable by only one of these groups, it will not be broadly implemented by the majority of persons who fall between these two extremes."

Another important concern with the use of biocriteria in environmental-impact assessment is that the cause of biological impairment is often difficult to infer from measures of biological integrity. Changes in biological integrity may be caused by one or more environmental stresses produced by any number of management decisions. Recently, multimetric approaches have been proposed to develop thresholds with biocriteria that may be useful in identifying different types of stresses (Barbour et al. 1995). However, it will likely prove difficult to develop fate or process models that can relate the impact of a particular farm-management decision to the biological integrity of nearby streams and lakes. So, while the environmental-impact-assessment model summarized in figure 1b has the advantage of using decision rather than test endpoints, a disadvantage lies in the difficulty of linking specific farm-management practices to perturbations in environmental integrity.

The EPA has been providing guidance to the states on the development and use of biological criteria (Southerland and Stibling 1995). Although at first glance biological criteria may appear complicated to implement in IPM assessment programs, IPM researchers and practitioners are already using biological indicators in their research on beneficial organisms and predator-prey relationships as indicators of community structure and trophic dependencies.

Spatial and Temporal Scales

In defining environmental impacts, it is important to consider a range of temporal and spatial scales, not just what happens on or near the farm in the current year. Usually, research focuses on localized small-scale, short-term impacts or on large-scale, long-term impacts, as illustrated by the diagonal line drawn in figure 2. However, off-diagonal processes are often important; for example, long-term effects of chemicals on the genetics of organisms or the rapid transfer of a chemical over relatively long distances through preferential flow.

Spatial and temporal scales are also important to consider when data are transferred between disciplines, when data are used to infer trends, and when data produced at one scale or in a narrowly defined system are used to interpret studies at a different scale or in a wider system, such as a landscape. Impacts of agriculture are generally experienced at spatial and temporal scales much larger than those at which environmental measurements are made. Processes in the landscape occur over a wide range of scales, but sampling is usually restricted to scales of time and space determined by sampling procedures and the time frame of a research or monitoring project. For example, soil scientists measure and monitor chemical concentrations at scales ranging from soil profile to field during experiments that rarely last more than a few years.

How should we approach measurement and monitoring at larger scales? Applying conventional measurement techniques to more sites for longer time periods can provide useful information, but it requires excessive effort and is costly. We need to rethink the way in which we approach such broad-scale projects, starting with an assessment of pathways and impacts and tailoring monitoring strategies to the whole system rather than to a few arbitrary points in it. Field monitoring and measurement strategies for broad-scale projects should be carefully planned and evaluated, taking into account both temporal and spatial variability. Techniques for parameter estimation, monitoring, and modeling should change as we move from point of application to catchment or to regional scales and should attempt to predict responses and impacts over decades rather than months.

Natural Resource Use and Sustainability

Another consideration in assessing the environmental impacts of agricultural production and distribution is in terms of resource use, both depletion of nonrenewable resources and consumption or transformation of renewable resources. Assessments of resource and energy use often are found under the rubric of energy or resource analysis, life-cycle assessment, systems analysis, or systems ecology (Cottrell 1955; Odum 1971; Cook 1976; Daly 1980; Pimentel 1980; Odum 1983; Helsel 1987; Hall, Cleveland, Kaufmann 1986; Fava et al. 1991, 1993; Guinee and Heijungs 1993, 1995; Daly and Cobb 1994; Schroll, H. 1994; Hall 1995). These assessments generally depend upon measures of the quantity and rate of consumption of resources and also upon abiotic indicators of physical changes in the environment.

Choices of agricultural pest-management practices may have long-term impacts on atmospheric and soil quality. For example, United Nations scientists estimate that methyl bromide, which is used primarily as a soil fumigant in agriculture, is responsible for 5 to 10 percent of the thinning of the stratospheric ozone layer. Thinning of the ozone shield is an indicator of physical change in the environment that has been related to human-health problems, to effects on nonhuman biota, and to marine and agricultural productivity (Allen et al. 1995; UNEP 1992, 1994, 1995).

On a global scale, fossil-energy resources are finite and nonrenewable, although their use has quite different economic and social ramifications as a cost of production in different political jurisdictions. Fossil energy is used in agriculture directly as a fuel and indirectly as embodied in farm machinery, transportation, pumped irrigation, synthetic pesticides, and chemical fertilizers. When quantities of fossil inputs are converted to energy units (such as calories, joules, and BTUs), it can be seen that the ratio of energy input to output in agriculture has changed significantly over time and with changing priorities and options in production and distribution. Fossil energy and electricity use on U.S. farms had increased more than sixfold between the turn of the century and the late 1970s when oil-price shocks spurred energy conservation throughout the economy. At peak usage in 1978, direct and indirect energy use on farms was equivalent to 5 percent of total U.S. energy consumption, while energy inputs to the entire food system (including distribution and processing) have been estimated at three to four times that amount. By 1990, however, energy productivity in agriculture had doubled from the minimum levels of the mid-1970s because of conservation, reduced acreage tilled, and greater use of diesel fuel, which delivers more mechanical energy per unit than gasoline (Cleveland 1995).

The significance of energy as an economic cost of production is, of course, recognized by growers, but we stress it here because energy analysis is a means of making a link between socioeconomic factors and environmental consequences. It is estimated that domestic sources of high-quality fossil energy will be depleted within the lifetimes of people who are now middle aged (Hall, Cleveland, and Kaufmann 1986). This will likely have serious, widespread ramifications on our environment and way of life, affecting the scale and location of agricultural production, the delineation of marketscapes and food systems, the demand for agricultural land and labor, the use of synthetic (fossil-based) pesticides and nutrients, and interest in promoting nonfossil-based alternatives in pest control and fertilization. Despite the relatively short time scale of these projected changes, we have seen stops and starts in developing policies and pricing systems that inspire more efficient use of these resources. Therefore, we suggest that evaluating the environmental consequences of the use of nonrenewable resources and slowing the use of renewable resources may provide additional insights and leverage in policy formation.

Summary: What is Environmental-Impact Assessment?

We consider the environmental impacts of agriculture to encompass all nontarget impacts, although in the context of the parameters mandated for this paper, we do not focus in great detail on direct impacts on human beings through occupational or other exposure. Nevertheless, it is important to realize that impacts on terrestrial, aquatic, and atmospheric systems clearly can have indirect impacts on human health; also that many of the nuanced, sublethal impacts that are being recognized on human health may have parallel impacts on nonhuman biota. We have attempted to show that many facets of the environment can be affected, directly or indirectly, by agricultural practices.

Environmental-impact assessments are measures or estimates of consequences of management decisions on one or more environmental indicators. They may be simply methods for identifying changes in the environment, or they may be tools for decision making that also assess the magnitude and significance of these changes.

Challenges in Assessing Environmental Impacts

In this section we shift from describing possible environmental impacts of agriculture to discussing some of the challenges and potential difficulties researchers face in developing systems to assess these impacts. These are conceptual challenges that are not, for the most part, likely to have quick technical solutions. The issues we discuss are organized into three sections: the identification and integration of environmental indicators; the bias against future impacts or, alternatively, our greater ease and ability in measuring and assessing current and tangible impacts; and the reality of data limitations that constrain the development of assessment models in covering the breadth of environmental parameters we mention in the first section.

Choosing Environmental Indicators and Deciding How to Integrate Them

As we have noted, many environmental indicators are needed to fully describe the environmental impacts of a pest-management product or method. To use the example of pesticide toxicity, there is no single species or group of biota that is most sensitive to all pesticides and thus useful as a surrogate for all others in toxicity testing. This truism applies to other environmental perturbations as well. We cannot rely on a single indicator species or abiotic effect to tell all we need to know about the impacts of any management decision. Scientists are therefore faced with the need to test and evaluate impacts on various groups of biota and then to integrate the results to create a composite assessment of environmental impacts of a pest-control method or other management strategy. One can grasp the conceptual challenge this poses by thinking about how one would go about weighting and summing an evaluation of impacts on human beings in relation to impacts on other biota, especially if the impacts were dissimilar in magnitude and type.

Another challenge to creating a composite assessment of environmental impacts of agricultural strategies is finding a meaningful common currency to describe different types of impacts. In answering many questions about environmental impacts, monetary values do not adequately describe non-market costs, such as the loss of an individual life, loss of biodiversity, impacts on nongame species, disruption of an ecosystem, future costs of current soil erosion, or loss of irreplaceable resources. Ongoing research in several disciplines is aimed at devising means of valuing environmental and other nonmarket goods; much of this work falls under the rubric of ecological economics (Daly 1991; Daly and Town-send 1993; Daly and Cobb 1994; Guinee and Hei-jungs 1995; Krishnan, Harris, and Goodwin 1995).

In some agricultural impact-assessment systems, both environmental parameters and on-farm economic costs are rated on a unitless scale; in others, on-farm costs are quantified in monetary terms, and environmental costs are indexed separately and 'flagged' to indicate a hazard or high risk. In a number of other systems, monetary values are imputed to a range of environmental impacts with one of several methods, such as replacement or remediation costs, lost productivity, or willingness to pay (contingent valuation) as the basis for assigning value to impacts. The drawback to remediation or replacement-cost accounting is that money is only a useful measure of impact if the environmental parameters or organisms in question are of intrinsic economic interest to people or if the costs of previous remediation efforts are known (see Pimentel et al. 1992). Contingent valuation is a useful measure only if the group surveyed for their willingness to pay are realistically able to assign monetary values to the nonmarket goods in question and are not swayed by thinking there will be possible economic or regulatory ramifications from answers that are biased high or low. Surveys to find out how much money individuals would be willing to pay for a nonmarket good are valid only when the sample represents the population that will bear most of the associated costs or reaps most of the associated benefits. To give an example illustrating this last point: a farmer's willingness to pay to avoid polluting water with a toxic pesticide or fertilizer runoff is not a reasonable or accurate way to value this environmental damage because all of society suffers from the results of such pollution and pays the costs of remediation. On the other hand, a survey assessing farmers' willingness to pay to avoid toxic risk to pesticide applicators may indeed be a reasonable method of valuation because this environmental cost affects farmers disproportionately. In designing assessment systems, it is important to remember that willingness to pay does not measure the existence or extent of an environmental problem; rather it measures attitude toward a problem and whether the problem bothers a particular stakeholder enough to pay for an alternative (Levitan et al. 1995).

Another challenge of creating composite assessments of environmental impacts is that no one set of social or environmental indicators is most appropriate to use in assessing impacts of agriculture. Different circumstances and objectives prioritize different indicators and interpretations. One may answer the question of how to integrate, weight, and value impacts in the context of one assessment scenario, but these issues will reemerge when the question of environmental impacts is asked on a different scale or with different objectives. For example, the types of data required to create a decision model for a farmer to use in the field in choosing a least-impact but efficacious pest-control method may not be the same as the data required for a national policy model assessing agricultural practices. To illustrate: while IPM farmers want to avoid using pesticides that harm parasites and predators specific to the crop pests in their fields, these producers might be misled by a decision model based on the more generic information about impacts of pesticides on beneficials that might be used in a national model of environmental impacts of IPM. Were the national model to consider impacts on beneficials at all, it would most likely rely on EPA data on acute toxic impacts of pesticides to honey bees, which are the only beneficials included in EPA's Ecological Effects data set (U.S. EPA 1996). Even if the toxic dose responses were comparable for honey bees and other beneficials, the significance of these effects might be quite different. When honey bees are repelled from a field by pyrethroid pesticides, for example, they survive and move on to another nectar source; however, if beneficial parasites and predators are repelled from a location, they are not then available to work as biological control agents. The design of an assessment system must, therefore, be appropriate to the objectives of the audience served.

Bias Against Future as Compared to Present Impacts

There are several ways in which we can be biased against considering future, as compared to present, impacts. Returning to our space-time diagram (fig.

2), the issues that tend to concern us most are those that occur in our immediate space and time frame. This implies that current activities that lead to environmental impacts at more distance places and times tend to receive less attention. For example, most ecotoxicity testing of pesticides emphasizes their short-term lethality rather than their chronic and cumulative impacts. Or we may be more interested in the short-term reduction in pesticide use that occurs when pest-resistant varieties are introduced than in the long-term impact on pest populations caused by the use of pest-resistant varieties. Long-term and cumulative impacts are more difficult to comprehend and quantify than short-term impacts, and less data are generally available. As a result, less weight tends to be given to these impacts in environmental assessments.

A second manner in which we can be biased against the future as compared to the present is by not considering impacts associated with future events (Garetz 1993), such as leaking of improperly stored pesticides in the future. Assessing future impacts of future events can be more uncertain than assessing impacts of current events, but this does not mean that such impacts are less important. For example, the Superfund Program and Hazardous Waste Program were established primarily on the basis of future rather than current risks.

Another problem for current assessments is that, as environmental systems change or become better understood in the future, the impact of IPM and other farm-management systems may be assessed differently. This assertion implies that assessors must be aware of new information and problems and be prepared to modify or change their assessment methods to account for changes in our knowledge base.

Data Limitations

Data are required at all stages of environmental assessment of agriculture. Data can be divided into different classes. Recognizing the variety of types of data enables us to place the availability of data into perspective. Data that describe intrinsic properties of a system are unlikely to change with time. Examples of these are soil data, rainfall, and climate records. Other data are valid for short time periods, such as farm-management information, and therefore have to be collected frequently. Yet other data may vary according to the type of assessment or as new knowledge becomes available. For these reasons, it is difficult to define a minimum data set for IPM planning and evaluation that will be widely applicable or remain constant for a long time. Because many environmental impacts are produced on different temporal and spatial scales than they are experienced, data for assessing these impacts cannot be collected on-farm, an important factor that differentiates environmental assessments from farm-scale economic assessments of IPM and other agricultural systems.

Toxicological- and ecological-effects data sets of pesticides are incomplete. In addition, some of the existing toxicity data are inappropriate to use as the basis for assessing relative impacts of different agricultural management strategies because they were not collected with standardized protocols and, therefore, are not comparable (Levitan et al. 1995). Moreover, there are very limited data and no standardized data sets on new biocides, such as microbial and fungal pesticides. The scientific community is only beginning to develop tools and to collect data for assessing positive and negative environmental impacts of biointensive IPM practices. The reasons for this are twofold. First, there are many interlinked physical, chemical, and biological processes that play a role in IPM, and it would be unusual for all of these processes to be fully understood and quantified for specific evaluations. Second, natural systems are inherently variable, both in space and time, and, to characterize both their average behavior as well as their variability, high-intensity sampling is required. Because it is often the occasional extreme occurrences that may lead to environmental damage, it is important to be able to predict the likelihood of these events (Wagenet and Hutson 1994; Jury and Gruber 1989).

As we note in an earlier section, most available data on pesticide environmental impacts originate from toxicity tests on single species of biota. In addition to limitations associated with testing single species of organisms, these data are also of limited value because the pesticides tested are generally applied in single doses of individual active ingredients.

Impacts to the environment, however, are from mixtures of active ingredients, whether tank mixes or mixes of residues in the environment, that can be greater or less than the sum of impacts from individual toxins. Cumulative impacts from repeated or extended exposures can also be different than impacts of single, larger exposures. Little is known about cumulative impacts and interactive effects, particularly in terrestrial systems, even though both human and nonhuman biota are virtually always exposed to chemical mixes and amounts that change spatially and over time (Yang 1994). Yang concludes that the toxicology of long-term, low-level exposures to chemical mixtures produces subtle effects, unlike acute toxic responses to higher doses; that such toxic interactions are possible at environmentally realistic levels; that the toxic responses may be from unconventional endpoints that are not usually tested; that there is a possibility that residual effects may become interactive with later exposures; and that these exposures may pose a safety risk to the public. While these comments are intended to apply to human subjects, we can extrapolate these principles and concerns to nonhuman biota, some populations of which may be more vulnerable to such risks because of limited mobility and physiological factors.

Summary: Challenges in Assessing Environmental Impacts

Although most of us support environmental-impact assessment in theory, many may express considerable skepticism about environmental-impact assessment in practice. There are numerous practical and theoretical problems in designing and conducting environmental-impact assessments. In this section, we have identified several challenges or concerns that can be raised in relation to most efforts at environmental assessment. We take the view that these are legitimate concerns that in many cases cannot currently be adequately addressed. However, we would argue that delaying environmental-impact assessment until these concerns can be dealt with effectively is not likely to be a productive strategy. Rather, environmental-impact methods are likely to be gradually improved as more researchers attempt to implement environmental assessments.

Methods for Impact Assessment

In this section, we review several categories of environmental-impact-assessment methods, including surveys and monitoring, fate models, and categorical indices of impacts. In each case, we discuss the objectives, strengths, and limitations of the methodology. All of these approaches have been used in environmental assessments of agriculture. The aim of this section of the paper is to encourage IPM researchers to actively consider the objectives and assumptions of the methods they are using and to refine methods, where feasible, rather than mechanically adopting methods without appropriate adaptations. In this way, researchers will not only increase the usefulness of their assessment, but may also contribute to the development of environmental-assessment methods.

Sampling and Monitoring

Of all the methodologies we will be discussing, sampling and monitoring are the most familiar to IPM researchers. Sample surveys are used in many fields to characterize populations (used broadly here to include biotic and abiotic phenomena) that are too large to census. Monitoring of various components of the environment usually involves repeating sample surveys over time. However, there are cases when monitoring involves measuring changes in the entire population of interest rather than in a sample of that population, for example when monitoring changes in a population of some endangered species. In any case, the major objective of monitoring is to address questions concerning the present status, changes, and future trends in the population that is being monitored (Larsen 1995).

On the national level, the U.S. Geological Survey, the USDA Soil Surveys, and the national network of weather stations have long been engaged in surveying the physical resource base of the nation and in providing this information to the public. More recently, there has been a growth in the use of surveys to characterize the natural and agricultural resource base. Examples include the National Agricultural Statistical Survey, the Forest Inventory Assessment, the National Wetlands Inventory, and the National Acidic Precipitation Program’s survey of lakes and streams. Surveys conducted over time add a temporal dimension to survey data, thus moving beyond a snapshot approach to resource inventory and essentially becoming a monitoring exercise. The EPA's Environmental Monitoring and Assessment Program (EMAP) is an example of a program designed to track changes in important environmental indicators that have been selected to characterize the condition of the nation's ecosystems. Another example of an environmental monitoring program is the Swiss National Soil Monitoring Network (Desaules 1993).

IPM researchers are familiar with sampling and monitoring of the environment at the local level because these activities are a major part of IPM research and practice. The strengths and weaknesses of surveying and monitoring are similar at local and regional levels. Surveys based on population samples make it feasible to characterize environmental resources, such as soil, lakes, and streams, as well as biotic populations that are too large to census. Otherwise, the status of a population would have to be inferred from an indicator or other species or simulation modeling. Monitoring can also be used to provide data for evaluating whether a system is changing and to predict future trends.

Obvious problems with sampling and monitoring are those of cost, convenience, and extrapolation. Often, so many samples must be taken to validly describe a population that the cost of sampling may become prohibitive. At other times, it can be impractical to choose a valid sample population. For example, farmers who are interested in working with extension agents and researchers to implement new pest-management strategies are not necessarily representative of the entire population of farmers who are using more conventional techniques. Given the voluntary nature of such arrangements, it may not be practical to select an unbiased sample of farmers. Lastly, without using other tools, the results of the sampling and monitoring work cannot be used to draw inferences about other populations (i.e., other farms, other practices, other components of the environment).

There are several other problems associated with monitoring beyond those of cost, convenience, and inability to extrapolate to populations not represented by the sample. Much of the rationale for monitoring lies in trend detection. However, in some environments, trend detection has been likened to looking for a needle in a haystack, with the needle being very small changes representing a trend lost in the haystack of measurement error and natural random fluctuations in time and space (Oliver 1993). Clearly, knowledge of natural fluctuations in time (e.g., seasonal effects) and space (e.g., soil types or soil depth) need to be considered in designing a monitoring system (Oliver 1993).

Dynamic simulation models can be used to predict temporal and spatial fluctuations and potentially to improve the design of a monitoring system. When the trend is very small compared to natural fluctuations in time and space, then other approaches need to be considered. An interesting improvement over standard monitoring is the combination of regional mass balances with monitoring data by the soil monitoring network in Switzerland mentioned above (Bader and Baccini 1993; von Streiger and Obrist 1993). The approach used in the Swiss study is to identify various categories of farms and then apply a model that distributes system inputs and outputs by farm category with regional average data. This method was used to identify agricultural land at high risk for copper contamination (in this case it was 11.9 percent of the total cultivated land) and then to focus monitoring activity on this smaller area of cultivated land at high risk. Such an approach can guide those responsible for monitoring and can influence how often and where samples should be collected.

Fate Models

Integrating and extrapolating physical, chemical, and biological processes in the environment is an essential part of assessing impacts of agriculture. Natural systems are dynamic. Models identify the relative importance of various dissipation pathways, and allow estimation of flux densities, concentrations, residence times, and exposure. Because most data collection is performed at detailed scales, simulation models are an attractive option for extending these data to broader space and time scales. Models may be viewed as repositories for dynamic processes, analogous to databases, which are often repositories for static data only.

Dynamic simulation models vary in their scope and complexity (Addiscott and Wagenet 1985), falling into broad use categories of education, screening, regulation, and research. The simplest of these models require few data and sometimes contain overly simplistic assumptions, but are easy to run and are useful for demonstrating the principles of environmental interaction. Screening models are usually used to rank chemicals in terms of potential environmental impact, and generally compare the relative impact of different chemicals against a constant environmental background. Models currently used for pesticide registration include environmental dynamics (rainfall, temperature, etc.) but exclude processes that may be important but are currently difficult to quantify, such as sorption kinetics. In regulatory models, processes are often represented as simply as possible, consistent with current knowledge and available data. Regulatory models make extensive use of libraries of existing databases and are structured to perform multiple executions easily. Research models are the most detailed in terms of their representation of processes. Their data demands are usually high, and considerable knowledge and experience are required to use them effectively.

The complexity and dynamic nature of environmental processes make simulation particularly attractive. The use of computer simulation models is increasing despite controversy over their validity and applicability. The controversy arises from opposing views of how models should be used. At one extreme are those who feel that models should contain only processes that have been proved valid and that they should not be applied outside a range of situations for which they are applicable. At the other extreme are those who would apply models even though the processes or data are known to be inapplicable to the situation under study. Useful applications probably lie between these two extremes, especially when combined with a critical and insightful evaluation of the output. Hauhs (1990) suggests that models should be applied until they are shown to be invalid, because they represent the current level of knowledge. However, if evidence from measurement, monitoring, or experience suggests that the model is deficient or inappropriate, then the scientific foundation of the model should be reexamined and improved.

When a model is used outside the situation in which it has proven applicable, it is important to remember that the model is a hypothesis and that subsequent measurement may prove it invalid or incomplete. Other approaches and available data should be reviewed before embarking on a modeling exercise. Such a review will highlight areas where there are insufficient data, thus highlighting the role of model output as a possible substitute. During this evaluation process, major mass-balance components may be estimated and deemed sufficiently accurate to satisfy demands of other disciplines.

Environmental evaluation often consists of the application of established scientific principles or models from several disciplines to larger-scale systems. The models employed at this larger scale are based on processes determined at the research scale. Processes that control responses at the larger (e.g., catchment) scale should be included but are not necessarily present in smaller-scale models. At larger, more complex levels, direct cause-and-effect relationships are more difficult to establish, and existing process-based models may become inadequate. Long-term experience and monitoring may become the sole measures of behavior at larger scales. But if models are viewed as providing hypotheses about system response at the larger scale, then it may be possible to design experiments or measurement exercises that can help assess the models. In this way we may develop a science at the larger environmental scale that does not depend completely on scaling-up of local-scale research.

Index or Ranking of Impacts of Pest-Control Products and Methods

Whereas monitoring systems tell you what is found at a particular time and place and fate models estimate what is likely to be found at other times and places, indexing or ranking systems for environmental-impact assessment estimate relative impacts of agricultural practices, such as the use of different pesticides. To explain this method, we describe a generic indexing system in which biologically or ecologically significant threshold levels for an environmental variable are used to define categories of impact, hazard, or risk. For example, if a certain pesticide kills half of a sample of honey bees at an exposure level less than one microgram per bee, that pesticide is categorized as posing a high risk to honey bees.

Some indexing systems use categories, such as high, moderate, low or no risk; in others these categories are analogous to the colors at a stop light: red for high hazard, impact, or risk; yellow, where there are moderate impacts and the practice should be used with caution; and green to indicate there is little or no impact from the practice. In some systems, these categories are scored, and the scores serve as the common currency to be weighted and summed in creating a composite assessment of impact from the practice. In other systems, continuous numerical ratings are used rather than discrete categorical interpretations of the data about impact. These numbers may be derived directly from toxicity tests (such as an LD_{50} value), may be a numerical test result modified by an exposure factor or other situation-specific property, or may be a ratio of environmental concentration to an effective concentration that causes a measurable impact (such as an LD_{50} or EC_{50}). In other systems, such as the World Wildlife Fund's assessment of adoption of IPM practices described by Hoppin (this volume Part II), the categories are behavioral. They are expressed as types of IPM practices (low-level, medium, and biointensive IPM) rather than as categories of magnitude of impact. In such behavioral systems, a relationship is assumed between certain behaviors or practices and the impacts of the practices.

Indexing and ranking systems are well-suited for comparing relative impacts of similar pest-management options, such as comparing toxicity of different pesticides, each of which has been assessed for the same endpoints at similar levels of exposure. Because of the conceptual difficulties in integrating different measures and indicators of impact, there is a greater margin of creative interpretation when indexing is used to compare impacts of quite different options. Some examples are comparing impacts of herbicides to control weeds versus tillage or comparing regional food-production systems where pesticides may be used to the environmental impacts of transporting organically produced food from a different agricultural region. Such systems are well suited for evaluation with hybrid assessment tools that draw on the strengths of both indexing and simulation methods.

Indexing systems are useful for evaluating many types of environmental variables, not only those that can be sampled, monitored, or mathematically modeled. It enables the leap from assessments based on test endpoints to the development of systems for assessing decision endpoints. We return to the example of the impact of different pesticides on honey bees to illustrate the difference: The measurement of toxicity to an organism is a test endpoint that provides data on the rate of pesticide application lethal to bees or the rate at which certain behaviors (such as nectar-collecting activity) will change. However, what a beekeeper is more likely to want to know is the combination of factors affecting hive survival or crop pollination. Management decisions of farmers and beekeepers could be affected by knowing how the impact on honey bees might be reduced by using a different pesticide, a lower dosage, or a different time of application.

In this example, acute toxicity to adult honey bees may not be the crucial variable for the beekeeper's decision because the most toxic pesticides may rapidly kill worker bees in the field or repel them from the field (as pyrethroid insecticides do), whereas somewhat less-acutely toxic pesticides may mix with the nectar or pollen and be brought back to the hive and fed to the brood, which is the next generation of workers. Or the less acutely toxic pesticide may have a sublethal impact on the adults, reducing their activity level and decreasing long-term chances of hive survival. Indexing systems have the potential of integrating test endpoints and ranking decision endpoints. A decision-making aid for determining whether a situation is hazardous to hive survival or pollination success might require the integration of a number of tests. Decision models for efficient and safe management practices for farmers, growers, livestock managers, and beekeepers might differ from each other and also be different from assessment models intended to summarize long-term and off-farm impacts to the environment and society. Without modifications (such as those described in this example) to incorporate site- and situation-specific factors, ranking systems reflect a generalized condition. In pesticide-ranking systems, site- and situation-specific factors include dose, time of day and season of application, and qualities of the formulated product.

A challenge in developing indexing systems is that the integration of impacts on specific endpoints into a composite assessment of impacts on the environment involves value judgment. The challenge is in justifying these judgments and in creating assessment tools that are sufficiently transparent and flexible to enable situation-specific modifications in the integrating algorithm. As methods are developed to incorporate situation-specific sensitivity to impacts, the value of indexing systems will improve.

Directions and Trends in Impact-Assessment Systems

We identify three areas in which we expect to see important changes in the development of impact assessment systems for agriculture:

1. More data must be produced on environmental impacts, broadly understood to include a range of environmental indicators. Perhaps it is even more crucial to stress that improved datasets of high-quality, comparable data (i.e., collected under standardized and recommended protocols) must be organized and made accessible to the assessment research community.

2. With better data and with a broader conceptualization of environmental impacts (going beyond single-species toxicity testing and measures of pollutant concentration in water), assessment systems will evolve to consider additional environmental variables and endpoints.

3. Developers of assessment systems will collaborate to overcome limitations of each individual methodological approach and will synthesize and build on the advantages of monitoring, modeling, indexing, and other methodologies. Systems will be developed that are more transparent and flexible in setting impact criteria, in determining which variables to include in the model, and in weighting relative importance of these variables in the system. With improved input data, and these other modifications, assessment models will be able to portray a more holistic picture of environmental impacts.

Choosing an Assessment Method

In this section, we consider some practical issues that face many researchers and that can ultimately have an important, if not decisive, role in determining the outcome of an assessment method. These issues include identifying the decisions, societal values, and assessment endpoints involved in the environmental assessment and factors to consider when selecting an appropriate model. The aim of this section is to encourage researchers to consider these issues explicitly before choosing an environmental-assessment method.

Identifying Decisions, Values, and Assessment Endpoints

Throughout this paper, we have emphasized that environmental-impact assessment has no single, well-defined method. In the first section, we emphasized that there are numerous environmental assessment endpoints of interest to various groups. In the next section, we raised questions suggesting that it is still not possible to conduct a complete (i.e., holistic) environmental assessment. In the third section we discussed the objectives, strengths, and limitations of some existing methods for environmental assessment of agriculture, pointing out limitations to each of these methods. How, then, should IPM researchers determine an appropriate approach to use in assessing the environmental impact(s) of the management systems they are promoting? Suter (1995) states that the selection of an appropriate environmental-assessment method that will lead to an informed decision must involve not only the assessors but also must be guided by an understanding of the public values involved in the decision. He suggests that selecting the appropriate method requires addressing four questions: (1) What is the nature of the decision? (2) What societal values are involved in the hazard to be assessed? (3) How can those values be operationally defined as assessment endpoints? (4) What combination of models, test endpoints, and other data will most efficiently provide an assessment of the assessment endpoints in a form suitable for the decision? In the next few paragraphs, we discuss these and other questions related to choosing a particular environmental-assessment method.

Before selecting an environmental-assessment method, it is critical to determine who is expected to use the assessment method and the information it produces. Is the information to be used by government agencies to assess policy impacts, or by growers to inform them of the potential environmental consequences of management decisions? Because many pest-management systems involve multiple decisions, IPM assessments potentially involve contrasting the impact of a range of decisions (the impact of the application of different pesticides, at different rates, at different times, and at different places) rather than just contrasting the standard use of a pesticide with no use of a pesticide.

There can be multiple societal values involved in estimating hazards of pesticide use. Excluding human-health concerns, farmers are concerned about the impacts of pesticides on beneficials and the inducement of pesticide resistance in target populations. Regulatory agencies are concerned with how farm-management decisions may impact benchmark values for pesticide levels in water and air. Other government agencies may be interested in endpoints that are important on a global scale and thus subject to international negotiations (Cairns 1995). Many in the general public are concerned with the impacts of pesticides on nontarget organisms, while environmentalists are also concerned with long-term, ecosystem-level impacts that may not be safeguarded by current standards. Scientists are concerned with potentially significant, unstudied impacts. Depending on the environmental values of the assessment developers and target audience, assessments of environmental impact of alternative decisions could be primarily focused on the short-term versus the long-term consequences and on site-specific versus regional or national impacts. Some groups may be interested in potential negative environmental consequences of proposed practices and want these to be compared to the environmental impact of standard production practices. Thus, the assessment or decision endpoints of most interest are likely to differ among different groups (Suter 1995). A quotation earlier in the paper (Hughes 1995) suggests that an environmental assessment of IPM should include assessment endpoints of interest to a broad spectrum of interested parties. Cairns (1995), in an article dealing with future trends in ecotoxicology, argues that ecotoxicological information will need to be more site-specific and produced more rapidly.

The implications of Suter's questions referred to at the beginning of this section are that only once the nature of the decision(s), societal values involved, and assessment endpoints are identified can the models, test endpoints, and data necessary to assess the endpoint be determined. As Suter points out, despite this ideal, most assessments have to rely on standard test endpoints available from existing toxicity data. These values generally are not the assessment endpoints. In this case, the role of the assessor must include tailoring the assessment to the decision. When considering use of an existing environmental-assessment tool, it is important to determine whether the assumptions and data used in developing the tool are appropriate to conditions or systems under which it will now be applied. For example, a pesticide hazard rating developed for apple orchards may not be appropriate for vegetable- or grain-crop systems. There may be a need for further measurements, and it may also be necessary to refine or further develop the assessment tool.

Choice of a Model

Choice of a model will depend on the reason for modeling (i.e., the questions we expect to answer). For example, a screening model may provide all the information required if the objective is merely to rank chemicals in terms of their potential for reaching groundwater. However, if a site-specific assessment is required, then data pertaining to that site and its weather have to be included, which necessitates a more complex model. In a scientific study of isolated and controlled processes, a simple model is likely to be successful, whereas more complex models that include many processes are required for large-scale simulations. Regardless of the application, an intelligent selection of a model requires the user to have a clear understanding of how well the processes included in the candidate models describe the processes likely to be important in the field.

At the outset, we need to recognize that the processes included in models are usually elucidated under highly controlled conditions. Interactions between processes and their behavior under changing environmental conditions are rarely studied, except in field experiments limited both in space and time. Thus, models are constructed to predict behavior under field conditions and to extrapolate processes to other soils and over longer times. Because it is impossible to measure everything, it is inevitable that models will be used to provide an extension of empirical knowledge.

Toward a Holistic Approach to Environmental-Impact Assessment of Agriculture

We will close by referring to the objectives reflected in the title of this paper: "Environmental-Impact Assessment: The Quest for a Holistic Picture," but with this quest modified somewhat by the conceptual challenges and technical limitations we have described. We have stressed the point that no single assessment system could include all of the environmental parameters we have mentioned and do so accurately at all scales of operation (from decisions made on a farmer's fields, to evaluating regional or watershed impacts, to national policy models, to planetary assessments). Nevertheless, in designing and implementing assessment systems, we believe it is preferable to *think about* the implications and ramifications of an agricultural practice on all of a system rather than to think only about a limited portion of the system while believing or implying that it is an assessment of impact on the entire system. We need to remember that environmental processes continue to occur even if they are not being monitored, sampled, or included in the assessment model.

In creating decision tools from assessment systems, we must think broadly about environmental impacts and develop methods for integrating environmental costs, public-health costs, social costs, and on-farm costs without losing valuable information about each set of issues. What this suggests is that both environmental impacts (nontarget costs) and farm-cost data (target impacts) need to be collected but analyzed independently. Conclusions from an analysis of the monetary costs of pest control should not influence or mitigate assessments of nontarget (environmental or social) costs. After all, environmental degradation and resource depletion resulting from a given practice do not decline because the economic costs of doing without a pesticide are high. Environmental impacts do not go away just because there are few alternative practices or products available. However, while the environmental assessment should not be mitigated by production-cost data, the *decision* about which production strategy to follow must, of course, weigh the information gleaned about on-farm costs as well as environmental impacts. These decisions should not be made in a black box. When the economic costs of environmental protection are high, society perhaps needs to consider whether and how to shift that economic burden from the farmer or the consumer to a larger group. In order to have this discussion, the methods and results of impact-assessment systems must remain visible (fig. 3).

So what can be expected from environmental-impact assessment systems? As we have implied, there are many ways to evaluate the environment and many ways to integrate a summary of impacts from specific agricultural strategies. We suggest that one of the greatest values of developing environmental-impact assessment systems is that they will facilitate rational social discourse about the effects, implications, and sustainability of agricultural production and marketing systems. It is our hope and prediction that good assessment systems will draw a broader group of better-informed parties into that discussion.

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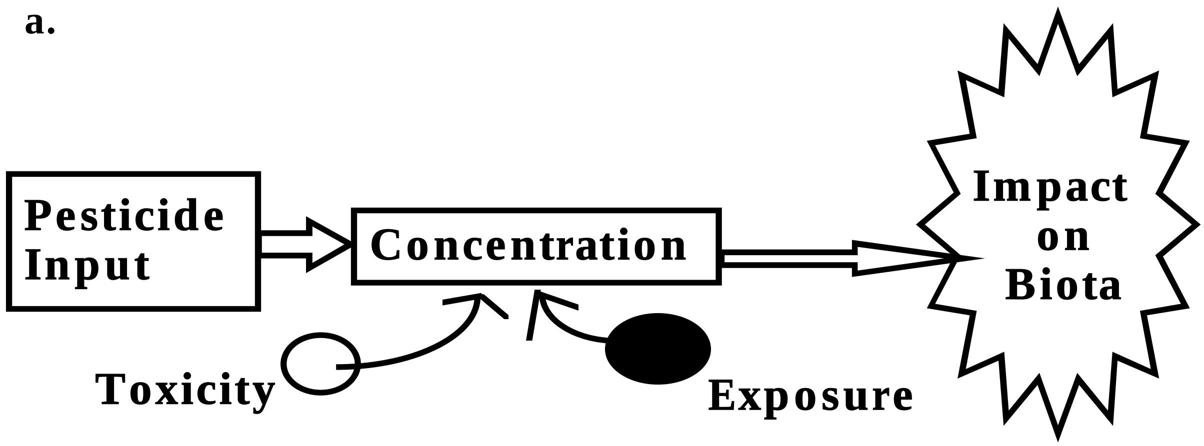
Figure 1a and 1b. Space and time scales of environmental studies.

a.



b.

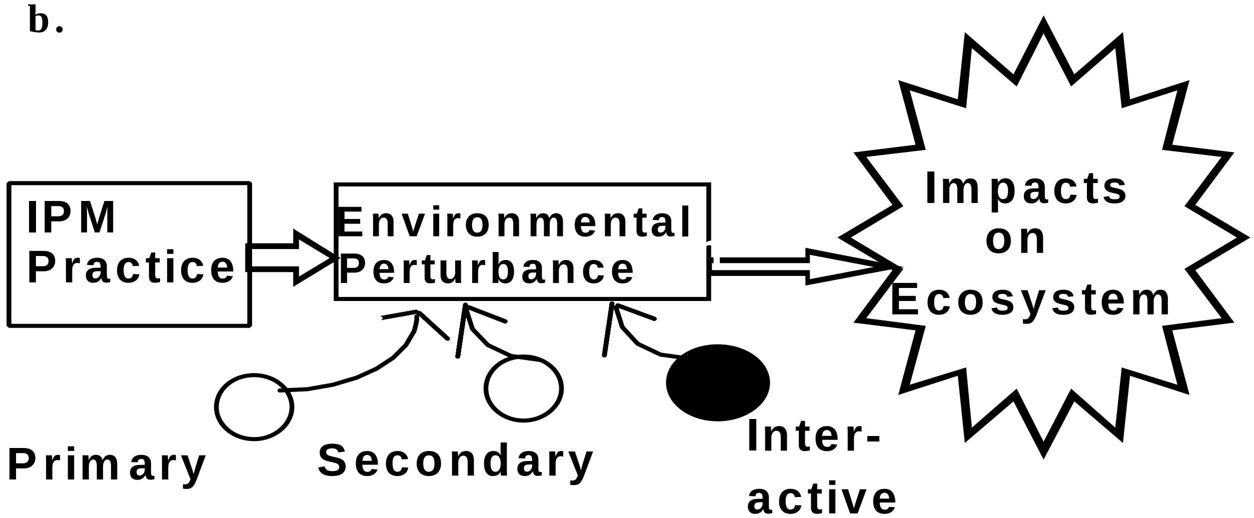


Figure 2. **Environmental impact models**.

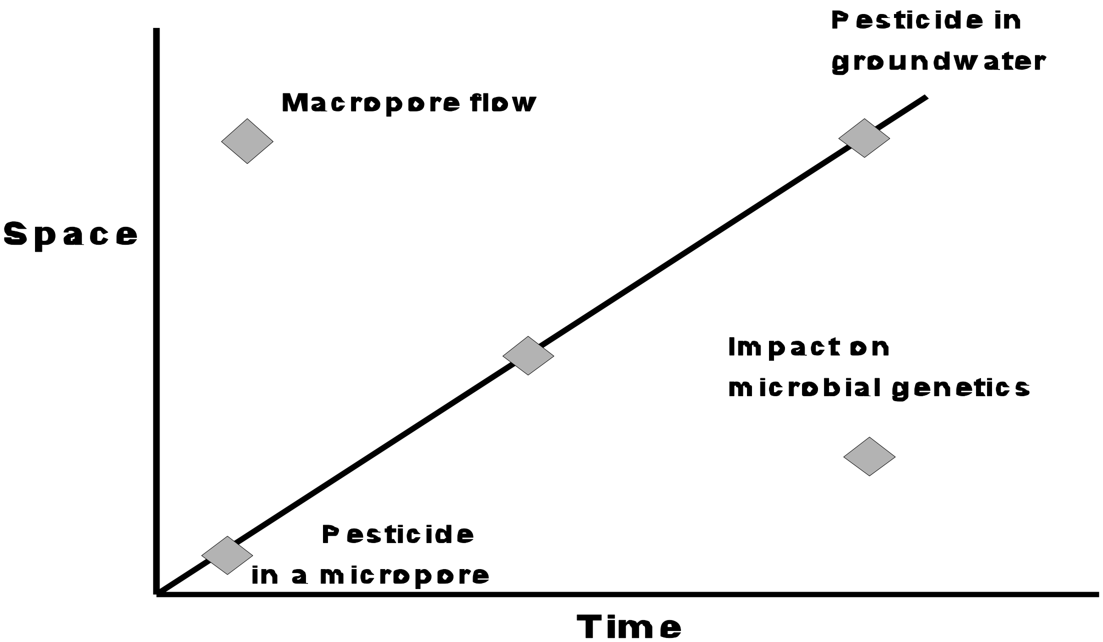
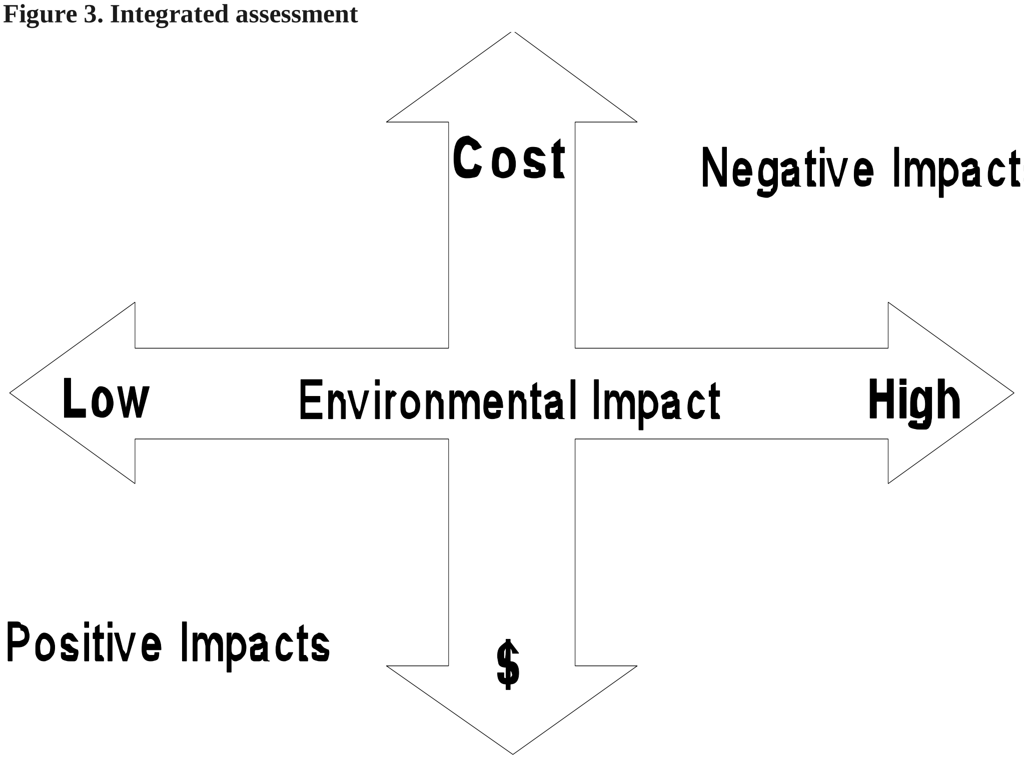


Figure 3. Integrated assessment



Occupational Exposures to Pesticides and Their Effects on Human Health

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Since the 1940s, use of synthetic pesticides has assumed an increasingly important role in control of pests in both agricultural and nonagricultural settings. Total use of pesticides in the United States has risen from an estimated 540 million pounds of active ingredient in the mid-1960s to 1,081 million pounds in 1993. Roughly three-fourths of this quantity is used in the agricultural sector, with the remainder divided somewhat evenly between home and garden and commercial and government use (USEPA 1994). The benefits of pesticides are many (Wilkinson 1990). On the agricultural side, they increase yields and diminish storage losses, thereby contributing to an abundant and inexpensive food supply. They have a direct role in public health through control of insects and other disease vectors.

While the benefits are substantial, there are costs associated with using pesticides. In fact, concern about potential human-health effects from these chemicals has paralleled their use and is usually credited with providing the stimulus for the environmental movement (Carson 1962). Modern industrial societies use many chemicals, but pesticides are unique in that they are designed to have adverse biologic effects. This property has accentuated the scrutiny they receive.

The adverse effects associated with pesticide use include impacts directly borne by the user, as well as those borne by society as a whole. Examples of the former include the development of pest resistance, secondary pest outbreaks, and damage to agricultural ecosystems. Examples of the latter include adverse impacts on worker safety, surface- and groundwater quality, biodiversity, ecosystem health, and consumer safety. These adverse effects can occur from direct contact with pesticides during mixing and application, from contact with contaminated equipment, from working the fields where pesticide residues occur, or from contamination of food or water. This paper will focus on the public-health impacts resulting from occupational exposures, but the other routes of exposure mentioned are also important, and discussions of these can be found in Nigg et al. (1990), NAS (1993), and Pease et al. (1995).

The challenge, then, is to strike a balance between the benefits and costs of pesticide use in agriculture. This is a difficult task given the complexities involved in detecting and monetizing many of the adverse impacts. But, as evidenced by the presentations at this workshop, there are emerging methods and approaches that can be used.

Integrated pest management (IPM) methods and techniques that diminish the frequency and amount of chemicals used, identify lower risk alternatives, and/or promote safe use and disposal of pesticides potentially could have measurable beneficial effects on human health. Identifying and measuring these impacts will require an understanding of the approaches and methods that public-health experts use to detect and measure the effects of pesticides on human health.

In this paper, previous research on pesticides and human health is summarized to highlight areas of concern about potential pesticide exposure and disease outcomes and to provide guidance for future research directions on pesticides. Results from epidemiologic studies are reviewed with a focus on chronic disease, particularly cancer. Possible mechanisms of action are discussed to provide a framework for research and evaluation of results. Techniques for monitoring pesticide exposure are reviewed to outline possible approaches for assessing changes in exposure associated with IPM techniques. Finally, approaches used in assessing public-health impacts are briefly described.

Assessing Human-Health Hazards

Three research approaches are currently used to obtain information on human-health hazards associated with pesticide exposure: (1) assessing links between exposure and disease, (2) relating exposure to biologic effects other than disease, and (3) evaluating exposure alone. These three approaches provide a hierarchical approach to research that focuses on different aspects of the exposure-disease process and that offers special opportunities in different situations.

The first category evaluates the relationship between pesticide exposure and disease. Pesticide exposure may cause acute and chronic effects. Chronic effects are much more difficult to evaluate than acute effects because years may pass between the initiating exposure and the development of disease symptoms. For cancer, the time period may be twenty or more years. This lengthy lag period creates many practical research problems, particularly the difficulty in assessing exposures that occurred many years in the past. Despite the practical difficulties, the approach focusing on the exposure-disease linkage is critical because it is essential to establishing a causal link and dose-response relationship.

The significant time lag between exposure and full-blown disease has been one motivation for the incorporation of laboratory techniques into human epidemiologic studies, particularly in cancer research. These new procedures are designed to evaluate the relationship between exposure to potentially hazardous chemicals and biologic effects that occur prior to full development of cancer or other diseases. Such a technique offers several advantages in our effort to understand environmentally caused disease. It greatly shortens the time between exposure and outcome because the period between exposure and many types of biologic damage is usually days or weeks instead of years, as with disease. This shortened response time occurs because the outcome of interest is not full-blown disease, but biologic damage or conditions that may eventually lead to disease. Examples of such biologic outcomes include chromosome aberrations, gene mutations, immune-system abnormalities, and hormone disruptions. Epidemiologic studies with laboratory components can also be very instrumental in expanding our general understanding of how diseases are caused. Such information can be helpful in developing new therapeutic procedures and interventions.

Direct monitoring of exposures is the third approach for assessing potential hazards posed by pesticides. It is the method of choice if there is already clear evidence that the chemical poses a hazard. In such situations, eliminating or minimizing the exposure is crucial. Exposure studies serve a range-finding function. If no exposure occurs, then obviously no hazard exists. Exposure studies also provide an indication of the appropriate level of concern because the toxicologic effect is usually proportional to the dose. Exposure studies also have a practical advantage over study of disease or biologic damage. For disease and biologic damage, some time must pass before assessment of hazardous effects is possible. With exposure monitoring, assessment is all that is required. This quick feedback has important preventive implications because corrective actions can be put into place promptly.

Human-Health Effects from Pesticide Exposure

Research on human-health effects serves as the basis for determining the need for preventive actions. Early research focused primarily on acute effects, such as poisoning; but more recently, interest in chronic diseases has increased.

Acute Effects

Although poisonings and death from acute pesticide exposures are well documented (Hayes 1975), statistics for most countries (including the United States) are incomplete. Given this caveat, there is some evidence that fatalities from pesticide exposure in the United States fell between the 1950s and 1970s (Hayes and Vaughan 1977). Information on pesticide-poisoning symptoms is even more limited than that for fatalities, and many symptoms undoubtedly go unreported or misdiagnosed. In California, where physicians are required by law to report pesticide poisonings, approximately 2,000 pesticide-related illnesses occur annually (Edmiston and Maddy 1987). A survey in Iowa in the 1990s found that approximately one-third of the farmers reported they had experienced some symptoms associated with pesticide use, such as headaches and vision difficulties (Blair et al. 1995).

Chronic Diseases

Chronic diseases are more difficult to evaluate than acute effects because they do not occur immediately after exposure. Some of the chronic diseases of concern include cancer (Blair et al. 1990) and diseases of the nervous system (Ecobichon et al. 1990), immune system (Thomas et al. 1990), and reproductive system (Mattison et al. 1990). The quantity and quality of the data available on these different diseases vary considerably. Cancer has received more attention than the others, and efforts are needed to correct this imbalance.

Neurologic Diseases. Diseases of the nervous system resulting from pesticide exposure are of special concern. Many insecticides target the nervous system of insects, thus it is not surprising that human exposures cause tremors, anorexia, muscular weakness, insomnia, convulsions, and depression (Echobichon et al. 1990). These symptoms have occurred with pesticides from a number of different chemical classes, including organochlorines, organophosphates, and carbamates. In a now classic study, many of the symptoms listed above occurred among workers with prolonged exposure to Kepone (chlordecone) in the Hopwell incident (Taylor 1985). In this incident, symptoms for many workers gradually disappeared after exposure ceased, but they persisted for several years in some of the most heavily exposed workers. Similarly, a study of individuals seeking health care for pesticide poisoning in California found they experienced neurobehavioral deficits (sustained visual attention and mood scales) and slower finger-tapping responses than individuals never experiencing a poisoning episode (Steenland et al. 1994). Recent studies of Parkinson's disease have suggested that pesticides may increase the risk of this chronic, debilitating, neurologic condition (Semchuk and Love 1995).

Cancer. The need to study human cancer and pesticide exposures is driven by several observations. First, pesticides were among the earliest chemicals evaluated for carcinogenicity in animal bioassays. To date, the National Toxicology Program has evaluated about 50 pesticides, and for about one-half of those tested there was some evidence of carcinogenicity (Huff et al. 1991). Carcinogenic activity occurred among pesticides in several chemical classes, including organochlorine, organophosphates, carbamates, herbicides, and fungicides. Although evidence of carcinogenicity in animals is not proof that the pesticide causes cancer in humans, positive bioassays do identify chemicals that need more intensive evaluation.

Epidemiologic studies of agricultural populations also indicate possible cancer hazards from pesticide exposure. In the 1970s the National Cancer Institute mapped cancer mortality rates at the county level (Mason et al. 1975). These maps provided clues for causes of cancer. The maps showed that many cancers clustered strongly in urban areas. For example, high lung-cancer rates were primarily located in the major metropolitan areas. On the other hand, for some of the lymphatic and hematopoietic cancers, high-rate areas were in nonurban, agricultural areas. Leukemia, for example, had a band of high-rate counties occurring in the central United States running from the Dakotas to Texas (Blair et al. 1980; Mason et al. 1975). These high-rate areas did not generally include cities and suggested that factors associated with the rural lifestyle may be involved.

Broad occupational surveys conducted in a number of developed countries provide information that can be used to evaluate mortality patterns among farmers. Overall, farmers are a very healthy group (table 1). Compared to the general population, they have a low overall mortality. Some of the diseases with strikingly low mortality rates among farmers include cardiovascular disease and cancers of the lung, esophagus, bladder, colon, and liver (Blair et al. 1992). In nearly every study, rates for total mortality; all cancer; and cancers of the lung, bladder, and colon were lower among farmers than among the general population. In terms of a healthy lifestyle, farmers are doing a lot of things right. Mortality rates for several of the cancers are low because farmers have a lower prevalence of smoking than the general population. Other factors that may contribute to lower risks include farmers' high level of physical activity and residence in areas with little air pollution.

In contrast to the generally lower mortality rates discussed above, farmers from many countries tend to experience elevated mortality from leukemia; non-Hodgkin's lymphoma; multiple myeloma (these are cancers of the blood and lymph system); skin cancer; and cancers of the lip, prostate, stomach, and brain (Blair et al. 1992) (table 1). Special death-certificate studies also found farmers experience excesses for these tumors (Blair et al. 1993). The tumors excessive among farmers do not fall into any obvious grouping other than they are not strongly associated with smoking. They vary in frequency, histology, and prognosis. The excesses for these cancers, against a background of low mortality from all causes, suggest a role for work-related exposures, and farmers have many potentially hazardous exposures, including pesticides. Several high-rate tumors among farmers are increasing in the general population, including multiple myeloma, non-Hodgkin's lymphoma, melanoma, and cancers of the brain and prostate (Devesa et al. 1987). Thus, understanding the factors contributing to these cancers in farmers may have broad public-health implications.

Mapping projects and mortality surveys suggest that farmers experience high rates for a few cancers. More sophisticated, analytic investigations are necessary to identify which, if any, factors in the agricultural environment contribute to these cancer excesses. Analytic studies at the National Cancer Institute have focused on lymphatic and hematopoietic cancers [i.e., multiple myeloma, non-Hodgkin's lymphoma, and leukemia (Blair and Zahm 1995)]. The strongest association identified to date has been between the herbicide 2,4-D and non-Hodgkin's lymphoma.

The studies mentioned above will be used to illustrate one investigatory method used to evaluate chronic disease risks from pesticide exposure. Investigations on non-Hodgkin's lymphoma in Kansas (Hoar et al. 1986) and Nebraska (Zahm et al. 1990) obtained information on the use of specific pesticides from interviews with farmers. Non-Hodgkin's lymphoma was associated with 2,4-D in both states, and relative risks (RR) rose with reported frequency of use. Farmers reporting use of 2,4-D 21 or more days per year had a relative risk of 7.6 in Kansas (table 2). Farmers who rarely used protective equipment, such as rubber gloves or masks, were at higher risk (RR 2.1) than those who used protective equipment (RR 1.6). Risk of non-Hodgkin's lymphoma also rose with frequency of reported use of 2,4-D in Nebraska to more than threefold among those reporting more than 20 days of use (Zahm et al. 1990) (table 3). In Nebraska, delay in changing clothing after applying 2,4-D increased risk of non-Hodgkin's lymphoma. Those who changed clothing right away had a relative risk of 1.1, those who waited until the end of the day had 1.5, and those who wore the same clothing the next day had 4.7. These findings indicate that simple protective practices, such as wearing rubber gloves and prompt changes of clothing, may be quite efficient in minimizing occupational exposure to pesticides during mixing and application. The associations between non-Hodgkin's lymphoma and reported use of the herbicide 2,4-D among farmers in Kansas and Nebraska could not be explained by established risk factors for this tumor or from use of other pesticides.

Not all studies evaluating non-Hodgkin's lymphoma and 2,4-D found an association. A study in Iowa and Minnesota found only a very small and statistically nonsignificant relative risk of 1.2 (Cantor et al. 1992). In this study, as in the investigations in Kansas and Nebraska, however, failure to use protective equipment tended to yield larger relative risks of non-Hodgkin's lymphoma from exposure to a number of pesticides, providing a further indication of the benefit of the safe handling of these chemicals.

Farmers appear to be taking more care while using pesticides. Preliminary results from the ongoing Agricultural Health Study of farm families being conducted by the National Cancer Institute, the National Institute of Environmental Health Sciences, and the Environmental Protection Agency show that, compared with 10 years ago, more farmers are taking protective actions during pesticide use (table 4). There is still room for improvement, but the trends are clearly in a desirable direction.

Immune System. The immune system acts to protect the body against foreign invaders. It is composed of a number of cellular and chemical components. Factors that affect the proper functioning of the immune system can have far-reaching effects and impact many diseases. Immunologic testing is relatively rare in humans, but a tiered scheme has been proposed for experiments in rodents (Luster et al. 1988). Few immunotoxicologic studies in humans have been conducted, but investigations in laboratory animals have noted decreased resistance to bacterial infection from methylparathion and carbofuran, decreased cytotoxic lymphocyte response from malathion, thymus atrophy from DDT, increased susceptibility to viral infection from dieldrin, suppression of T-cell activity from chlordane, and enhanced T- and B-cell immune response from 2,4-D (Thomas et al. 1990).

Reproductive System. Testing of pesticides for reproductive effects is far from complete. Chemicals appear to affect reproduction by direct germ-cell destruction or hormonal actions (Mattison et al. 1990). Some effects are known in humans. In men, the pesticide dibromochloropropane (DBCP) causes a decrease in sperm production and/or production of abnormal sperm (Milby and Whorton 1980; Lipschultz et al. 1980), while chlordecone reduced sperm motility (Taylor et al. 1978). DDT, methoxychlor, chlordecone, and Lindane have reproductive effects in animals, but effects in humans have not been carefully evaluated (Mattison et al. 1990). There is a need to develop and apply standardized techniques to evaluate potential reproductive effects of pesticides in humans.

Current Research

Several large-scale research efforts are under way to evaluate risk of cancer and other diseases among farmers and farm families from various agricultural exposures, including pesticides. In the United States, the Agricultural Health Study, a collaborative effort involving the National Cancer Institute, the National Institute of Environmental Health Sciences, and the Environmental Protection Agency, is designed to evaluate cancer, neurologic disease, and reproductive outcomes among 75,000 farmers, farmers' spouses, and children in Iowa and North Carolina (Alavanja et al. 1995). In this prospective investigation, information on pesticides obtained includes specific chemicals used, timing and frequency of use, and protective practices employed. The cohort will be followed for 10 or more years to identify diseases that occur. Participants will be recontacted periodically to obtain information on any changes in pesticide practices, including use of IPM practices.

In Canada, persons identified as engaged in farming from the 1970 Census were identified and linked to the Agricultural Census to obtain more information on their agricultural practices. This large cohort, which includes essentially all the farmers in Canada, will be followed to determine cancer incidence and mortality (Wigle et al. 1990). Analyses to date have observed associations between the use of herbicides and development of non-Hodgkin's lymphoma (Wigle et al. 1990) and prostate cancer (Morrison et al. 1992). Continued followup of the cohort for mortality and cancer incidence will allow the evaluation of risks of many diseases in relation to pesticide use and the production of various agricultural commodities.

In 1990, Congress provided the National Institute for Occupational Safety and Health (NIOSH) with special funding to initiate a program in agricultural safety and health. The program consisted of several components, including: (1) a survey of farm-family health and hazards to develop more complete information on disease and injuries among farmers, (2) research into etiology of diseases and injuries, (3) efforts to develop and improve intervention strategies, (4) surveillance to monitor results, and (5) cancer control demonstration projects (CDC/NIOSH 1992).

The National Cancer Institute is conducting a series of methodologic projects to obtain information necessary to plan epidemiologic studies of migrant and seasonal farm workers (Zahm and Blair 1993). This population of agricultural workers, despite opportunities for considerable exposure to pesticides, has rarely been included in epidemiologic investigations. Pesticide exposure at an early age and lack of facilities for cleanup may put migrant and seasonal workers at high risk of disease.

Biologic Effects of Pesticide Exposure

Incorporation of laboratory (i.e., biochemical) techniques into epidemiologic studies offers opportunities not available with more traditional methods. These new techniques can be instrumental in the investigation of many acute and chronic diseases (Schulte and Perera 1993), but they have been especially beneficial for cancer (Perera and Santella 1993). These biochemical measures can sometimes, but not always, be used to evaluate exposure from pesticides, mechanisms of cancer causation, and the relationship between exposure and biologic damage. Evaluations can be made more quickly than with the more traditional disease-related epidemiology and with small numbers of subjects. Disadvantages include a lack of a reliable and accurate laboratory procedure to measure dose or outcome and cost. Each test can be quite expensive.

It is possible to measure levels of a number of pesticides, or their metabolites, in blood or urine (Saleh et al. 1994). Biologic measures of exposure will be discussed in greater detail in the section on exposure assessment.

Research on cancer can be used to illustrate the benefit of biologic markers in the investigation of pesticide exposure and mechanisms of carcinogenicity. Pesticides may cause cancer or other diseases through several mechanisms, including direct damage to genetic material (e.g., gene mutations), damage to other important biologic molecules, or hormonal effects.

A number of pesticides are genotoxic (i.e., they cause genetic damage). In one study, genetic damage from 65 pesticides was evaluated through 14 different tests. About 50 percent of the pesticides showed some genetic activity. Nine pesticides were active in most tests, 26 were active in several tests, and 30 were inactive in all tests (Garrett et al. 1986). Chromosome damage (Garry et al. 1989) and genomic instability (Kirsch and Lipkowitz 1992) have been noted among insecticide and fungicide applicators in the grain industry. These findings indicate that pesticides may cause disease by directly damaging the genetic material, and this offers an opportunity for short-term evaluation of persons exposed to pesticides.

As we have noted earlier, pesticides may affect the proper functioning of the immune system, and this may have repercussions on a number of diseases. Pesticidal action through this mechanism also offers an opportunity to evaluate short-term effects of exposure. Newcombe et al. (1992) have proposed that organophosphate pesticides may play a role in carcinogenesis through their inhibition of certain enzymes (i.e., serine esterases). These enzymes perform a critical role in the proper functioning of T lymphocytes and natural killer cells in the blood. These cells, if functioning properly, destroy virus-infected and transformed cells that may be precursors for malignant lymphomas. Anything that affects serine esterases could, therefore, increase the risk of lymphoma, and some organophosphate insecticides appear to have this capability (Newcombe et al. 1994). A possible effect of organophosphate insecticides on lymphomas is especially interesting given the excess of this cancer often observed among farmers (Blair et al. 1992).

Recently concern has arisen that some pesticides and other chemicals may cause disease because they mimic important hormones (McLachlan 1993). Chemicals that have been shown to exhibit weak estrogenic properties include polychlorinated biphenyls, DDT, and Kepone. The theoretical basis for the action of such chemicals is that they mimic a hormone by binding to the hormone receptor molecule. Through this binding, they can elicit normal hormone actions, including reproductive, developmental, and carcinogenic effects.

The concern over chemicals with potential hormonal effects has been reinforced by recent studies of breast cancer. Several investigations have found higher levels of DDT, or its major metabolite DDE, among women with breast cancer than among women without cancer (Falck et al. 1992; Wolff et al. 1993). DDT is fat soluble and persists for years, even decades, in body tissues. Because of this persistence, measurements of DDT/DDE in blood provide an excellent indication of dose. This methodological approach of comparing levels in persons with and without a disease can be used for other chemicals that have long biologic half lives, such as other organochlorine pesticides.

Human-Exposure Assessment

One of the goals of IPM is to reduce the use of chemicals that are toxic to humans and the environment. It may be necessary to balance the use of greater quantities of less toxic products with smaller quantities of more toxic chemicals and to strike a balance between potential human-health risks and risks to the environment.

Human exposures to agricultural chemicals may occur through several routes. Pesticides may be inhaled during mixing, loading, and application or through volatilization or spray drift. Dermal exposures occur from direct contact with pesticides (concentrated or dilute) or with surfaces (e.g., equipment, leaves, and soil) that have been treated. Pesticide-contaminated soil or plant material may be blown through the air or tracked into the house. General environmental exposures may occur from consumption of pesticide-treated foods and drinking water that contains agricultural chemicals.

With varied routes of exposure, there are also many potentially exposed populations. One obvious group is agricultural workers who mix, load, and apply pesticides or who enter pesticide-treated fields. The families of agricultural workers may incur exposures from activities in treated fields, drift from application, pesticides tracked into the home, or by contact with contaminated trucks or other equipment (Simcox et al. 1995).

Exposures to the general public may occur from home pesticide use, whether it is applied by the homeowner or by a professional applicator, or from treated public areas, such as roadways and recreational areas. The EPA has sponsored a large nonoccupational pesticide exposure study (USEPA 1990; R.W. Whitmore et al. 1994). In addition, the general public may be exposed to pesticides from consumption of food containing pesticide residues or from contaminated drinking water. Of particular concern, following the National Academy of Sciences report *Pesticides in the Diets of Infants and Children* (NAS 1993), are exposures to sensitive populations, including the young, elderly, and immunocompromised.

To assess exposures in any of the above populations, accurate and reliable monitoring procedures are essential. There are many methods for measuring or estimating exposure to pesticides and agricultural chemicals. The types of exposure-assessment methods chosen depend upon the time and resources available.

Quantitative Exposure-Assessment Methods

Quantitative exposure-assessment methods have been used for decades for estimating both dermal and inhalation exposures to various occupational groups and are now being applied to other potentially exposed groups (residents, children, etc.). Measurement of exposures that occur via the dermal and inhalation routes will be the primary focus of this discussion. The EPA provides exposure-assessment guidelines for measurement of applicator and reentry exposures and for exposure assessment in general (USEPA 1987; USEPA 1984). These documents and the new Occupational and Residential Postapplication Exposure Monitoring Test Guidelines (USEPA 1996) provide a good background on various quantitative exposure-assessment techniques.

The measurement of pesticide residues in food, combined with a knowledge of the type and amount of foods we consume, is the most common method for estimating dietary exposure and will not be discussed here. There is software available for the calculation of dietary exposure (for example, TAS EXPOSURE I® and IV®). A more detailed discussion of the assessment of risk from food or water consumption is beyond the scope of this paper. The interested reader may find the following publications helpful, Chaisson et al. (1991), USEPA (1992), and NAS (1993).

Dermal Exposure. Dermal-exposure-assessment techniques estimate the amount of product that ends up on the skin during and following various tasks and activities. Generally, these methods require the collection of a sample that then undergoes laboratory analysis. Sample collection requires the availability of accurate and precise analytical methods for the chemicals of interest.

One of the simplest methods for determining dermal exposure uses patches on various body parts. A patch is generally a 2.5- to 4-in. square of cellulose, gauze, or some chromatographic material that is secured to the outside of clothing or hats. After exposure, these patches are carefully removed, packaged, and sent to a laboratory for analysis. Patches are generally placed on the head, tops of the shoulders, on the back of the neck, on the upper chest, in the back of the forearms, and in front of the thighs and lower legs. It may be necessary to place additional pads depending upon the work task and the clothing worn. Patches may also be placed under the work clothing to estimate the amount of product that penetrates through the material.

A more accurate estimate of total-body exposure can be made if entire garments worn during the task are removed and analyzed for the chemical of interest. These commercially available garments must be removed carefully to prevent cross-contamination. It is possible to extract chemicals from the entire garment; however, generally, the garment is cut up, and individual segments are analyzed. This allows the estimates of exposure to arms, trunk, and legs to determine which body parts receive the highest exposures.

Unprotected hands have the greatest potential for dermal exposure. Even when protective gloves are worn, products may penetrate the gloves, or pesticides may be transferred to the hands when the gloves are adjusted or removed. Historically, the method for measuring hand exposure is the hand rinse. After exposure, hands are rinsed in a solvent to remove the pesticide. Isopropanol is commonly used; however, other solvents, including water with a surfactant, may be more appropriate, depending on the chemical of interest. The person exposed may wash his hands in a measured quantity of solvent in a basin, and the washing solution is collected and analyzed. Alternatively, a person places his hands in a plastic bag containing a measured amount of solvent and shakes his hands for at least 2 minutes. The bag is then closed and sent for analysis. This method is simple but highly variable (Fenske et al. 1994) because it is difficult to remove all pesticide from the hands, particularly around the fingernails and cuticles.

Sampling gloves may be used for estimating the total hand exposure. These gloves may be worn alone or inside of work gloves. Generally, these gloves are made of cotton (pall bearers' gloves) or of nylon knit (pickers' gloves). The nylon knit is stronger and less likely to rip or be punctured during normal work tasks. The gloves are peeled off so that they are turned inside out to prevent cross-contamination. As with the whole-body dosimeters, they are then sent to the laboratory for extraction and analysis.

A technique that may be applicable to certain liquid pesticide products uses a fluorescent tracer dye added to the tank mix for products that are sprayed. The tracer dye glows when viewed under ultraviolet light. Richard Fenske at the University of Washington has developed a quantitative method for estimating the amount of fluorescent material on the skin with video-imaging techniques (Fenske et al. 1986). This technique will not work for all potential exposures because of degradation of the fluorescent dye over time and with exposure to the sun. Also it is difficult to add the dye to some formulations. Fluorescent tracers, even without the video-imaging, show which body parts have been exposed to pesticides. This technique is an excellent teaching tool for showing workers how their activities and habits affect dermal exposure (Fenske 1988; Fenske 1990).

Inhalation Exposure. Vacuum pumps are used for measuring the quantity of a product in the air, either as a vapor or as an aerosol. The pump draws air through a collection medium. Small pumps can be worn by the person to measure personal exposure or it may be placed in the area to provide a stationary measure of exposure. Collection media for gases and vapors are usually some type of adsorbent, such as charcoal or chromatographic materials, or it could be a liquid solution that traps or reacts with the chemical of interest. Aerosols (particles or droplets) are generally collected on some type of filter medium or are trapped in a liquid. Filters are generally made of cellulose, glass fiber, or some type of plastic, such as PVC or polyurethane foam, and trapping solutions may be organic solvents or water-based weak acids or bases. The collection media are sent to a laboratory for analysis. It may also be possible to use direct-reading instruments in which a pump draws contaminated air past a sensor or into a portable chromatograph. This type of measurement technique provides for instantaneous assessment of exposure and is useful for education of the exposed person.

Respirators with an absorbent material in front of the filters represents an older technique to measure inhalation exposure. Quarter-, half-, or full-face filtering respirators may be used. The person wearing the respirator, in the process of inhalation, acts as the vacuum pump to draw air through the filter. This method provides a direct measure of inhalation exposure and does not require an estimate to be made about the breathing rate of the exposed individual.

Biologic Monitoring. Air and dermal sampling measure exposure at the person–environment boundary. To estimate absorbed dose from the measurement techniques above, assumptions must be made about the breathing rate and the amount of chemical absorbed through the lungs and skin. Measurement of chemicals or their metabolites in biologic media, however, can directly determine the amount of chemical that actually enters the body and integrates the exposures from all routes that occur over time. Care must be taken to collect the sample at a biologically relevant time period. Many pesticides are eliminated from the body in a few days; thus, the sampling must occur in close time proximity to exposure. See *Biological Monitoring for Pesticide Exposure* (Wang et al. 1989) for reports of various pesticide studies that used biological monitoring.

Urine is the most common, noninvasive, biologic medium that may be analyzed for pesticides or their metabolites. It is collected in a sterile container over a certain time period (usually 2 to 24 hours). The use of urine as a measure of exposure is based upon good toxicologic and chemical knowledge of the substance under study. Urine may not be the most appropriate medium if the metabolites are not specific, the substance is fat-soluble, or an analytical method is not available. One difficulty that may arise is that workers or other study subjects may refuse to provide urine samples because of concern about drug testing. Care must be taken to provide adequate information to the subjects concerning the purpose of the study.

Blood, plasma, and serum measurements are commonly used for the assessment of certain chemicals. For example, cholinesterase levels in the blood are an indication of exposure to organophosphate and carbamate pesticides (Hayes et al. 1980). However, this technique is invasive, requires trained personnel to draw blood, and is frequently opposed by the exposed person because of concern about possible infection.

Exhaled air may be collected to measure exposure to certain volatile and nonpolar pesticides. This technique has been used primarily for fumigants and provides a measure of recent exposure. Because it is noninvasive, it may be more acceptable to the subjects. Unfortunately, it is not always simple to get reproducible results. This technique is more useful simply as an indicator of exposure and not as a quantitative technique.

Surface Contamination. In addition to measuring dermal exposure directly, techniques for measuring the amount of pesticide on various surfaces are often valuable. An estimate of exposure may be made if the amount of chemical on the surfaces is known along with an estimate of the amount of surface contacted, the amount of material transferred from those surfaces, and a measure of dermal absorption. One method for determining the amount of dislodgeable foliar residue is to punch out circles from leaves or, for plants with small leaves, blades, or needles, by cutting representative samples. Pesticide residues are dislodged into an aqueous solution, usually a wetting agent in water. A second method for the collection of surface residues works well on turf or on surfaces like floors or carpets. This method involves dragging or rolling a sample-collection medium across the surface. The amount of residue on the collection medium and the area of surface contacted allows the calculation of the dislodgeable residue on that surface. The dislodgeable residues on hard surfaces may be measured by wipe sampling. An area of specific size is wiped across the area with an even pressure.

Two less commonly used techniques of surface sampling may be appropriate for certain conditions. A vacuum cleaner may be used to collect pesticide-containing dusts from hard surfaces, carpet, and upholstery (Lewis et al. 1994). Alternatively, in an experimental study, representative pieces of various household materials may be placed in the area before pesticide application. These coupons would then be removed and extracted or wiped.

Soil may also be sampled by removing soil samples from the surface and separating the soil into particle-size fractions. Generally, only particles less than 147 μm in diameter are extracted and analyzed for pesticide residues.

Exposure Models and Databases

As an alternative to the collection of air, dermal, and surface concentration data, a variety of models and databases are available for estimating pesticide exposure. Probably the most well-known database is the Pesticide Handlers Exposure Database (PHED). This database was developed by EPA, Health Canada, and the American Crop Protection Association. It consists of thousands of replicates of exposure data on mixers, loaders, applicators, and flaggers. Each replicate contains the measured dermal and/or inhalation exposures and the exposure factors that describe that particular situation including the type of formulation, amount handled, concentration, weather conditions, mixing/loading or application equipment, and crops or areas treated.

PHED is not chemical specific. The theory behind this database assumes that the formulation is the best indicator of exposure and physical and chemical characteristics of the pesticide are less important. Based upon this hypothesis, a database was developed along with various statistical and exposure-calculation software to allow an exposure calculation based simply upon the product use. For example, if one wanted to estimate the exposure of an applicator to a pesticide with an emulsifiable concentrate formulation that was applied in a specific amount via closed-cab air blast to peaches but had no actual measurements, PHED would provide both a dermal and inhalation exposure estimate. This model is a stand-alone program. Persons may also add their own exposure data or compare their data to that already in the database.

In addition to PHED, two European models exist for estimation of mixer, loader, and applicator exposure. The U.K. Predictive Operator Exposure Model (POEM) and the German BBA model use exposure factors for various formulation and application scenarios. Both models are available as EXCEL© spreadsheets. Comparison of the results of these two models indicates that POEM is generally more conservative than the BBA model.

Two additional databases are in the development stage. As a result of EPA data call-ins, industry groups have formed three task forces. There is a spray-drift task force that is developing data and models for spray-drift exposures. In the initial stages, the Agricultural Re-entry Task Force (ARTF) and the Outdoor Residential Task Force are collecting data and commissioning studies that will result in a database/model similar to PHED.

Advantages and Disadvantages of Exposure Estimation Methods

Quantitative exposure-assessment methods that involve the actual collection of air, dermal, or surface concentration data provide the most detailed and appropriate exposure estimates. They are chemical specific and exposure-scenario specific. Unfortunately, they are always expensive and involve time for planning, execution, and analysis. A worker-exposure study involving 15 replicate measurements may cost $100,000 to $500,000. Although the exposure measurements may be collected over a week, the preparation, analysis, and report writing may take a year or more. These studies depend upon the cooperation of the persons being monitored, which, if the exposures require the collection of biological samples, may be difficult to obtain.

Models and databases provide a good alternative. Unfortunately, these data are available only for pesticide mixers, loaders, and applicators. Other databases are being developed but are not yet ready for public use. The advantages to using models such as PHED, POEM, and BBA are that they are ready now and can provide answers quickly at little cost. The major disadvantage is that not all formulation/application scenarios are covered by these models. There are very little data for newer formulation types, such as the microencapsulated products.

Semiquantitative methods are useful for answering the present/absent exposure question but may not be appropriate when it is necessary to choose between two products. The detail of a quantitative exposure assessment is missing. Also, there may not be data available for the exposure conditions of interest (e.g., tracking a pesticide into a home).

Exposure Issues for IPM

The exposure-assessment methods described in this paper will allow the estimation of exposure, and with knowledge of the epidemiology and toxicology of the chemicals, human-health risks may be determined. Factors that play a critical role in the exposure calculation are the potential routes of exposure, the populations potentially exposed, and the amount of chemicals used. Exposure estimation may be simple or detailed, depending on the level of specificity of the answer that is needed. One of the most difficult aspects of exposure assessment is the determination of all potentially exposed groups. Frequently, only worker exposure is considered. Other populations that should be considered include farm families, bystanders, and persons who contact pesticides outside of the agricultural environment. Quantitative measurement of exposure is time consuming and costly. It is, however, precise and represents the situation of interest better than any other method. The use of exposure models and databases may provide quick, relatively inexpensive answers to exposure questions if the databases have information on a specific product and use scenario. If detailed information is not necessary, information from use records, pesticide registrants, and the literature may be sufficient for a gross exposure assessment.

There are a number of excellent researchers capable of providing information and guidance on quantitative and qualitative exposure-assessment techniques including the well-known academic scientists Richard Fenske at the University of Washington in Seattle, William Poppendorf at Utah State University in Logan, and Herbert Nigg at the University of Florida in Lake Alfred.

In addition, most of the large pesticide-manufacturing companies have industrial hygienists and regulatory toxicologists on staff who regularly perform exposure studies on their products. Pesticide manufacturers provide a starting point for the determination of what types of studies have already been conducted to assess exposure to their products. In addition, many private consulting firms specialize in exposure assessment to pesticides and agrochemicals.

Public-Health-Impact Assessment

Assessing the impact of changes in pesticide exposure levels and risk resulting from the use of IPM practices requires an understanding of the potential tradeoffs between risks to human health, environmental quality, and agricultural-production possibilities. How particular sets of IPM practices and technologies change pesticide-exposure levels and risk to the applicator, applicator's family, and other farm workers is a critical piece of data needed to assess these tradeoffs. However, exposure levels alone do not provide a comprehensive picture of the changes in risk to those in agriculture or society as a whole because pesticides can have multiple impact dimensions that include not only occupational health and safety, but water quality, wildlife habitat, biodiversity, and agricultural production, to name a few.

Public-health impacts must be incorporated into an integrated-assessment framework that facilitates the comparison of impacts of IPM practices on risk in other vectors of concern. Failure to assess changes in relative risk in a comprehensive fashion might result in a small reduction of risk in one vector and a large increase in another, resulting in a net increase in risk to society (Levitan et al. 1995; Mullen 1995). Methods used by economists and other environmental scientists to conduct assessments that include these multiple impacts are described in detail in this volume by Norton, Riha et al., and Antle and Capalbo (see also Mullen 1995; Levitan et al. 1995).

Estimating the monetary costs of real or potential public-health impacts is an important component of an integrated assessment. Several different approaches have been used to assess the public-health impacts of changes in production practices that reduce pesticide exposure. In cases where the dose-response relationship of a pesticide and a particular health outcome is established, a "cost-of-illness" approach can be used. By estimating the medical costs of treating the health outcome and the value of lost wages resulting from the illness, an estimate can be made of the health costs of using a particular chemical (Crissman 1994; Antle and Pingali 1994). The cost-of-illness approach represents the lower bound of estimated health costs. A more accurate measure of health costs would include an estimate of what people would pay to avoid becoming ill and the value of the suffering and inconvenience of being ill. Estimates of this "psychic" value can be obtained through surveys that ask people how much they would pay to avoid this adverse health outcome (Cropper 1994).

An example of the cost-of-illness approach is found in Antle and Pingali (1994). The authors found that for certain rice producers in the Philippines, when treatment costs and lost wages were incorporated into an overall economic assessment, the positive production benefits to the farmer from using the pesticide did not exceed the costs. In cases where, after incorporating direct health costs resulting from pesticide use, the cost of using that pesticide do not exceed the production benefits to the producer, then it is not necessary to estimate the psychic costs. This represents a "win-win" situation because productivity does not decline and risk is reduced. In cases where the production benefits exceed the costs, even with health costs incorporated, the value of avoiding illness must be incorporated (Cropper 1994).

In many cases, however, the dose relationship between a pesticide and particular health outcome is not clearly understood or quantified. Thus, it is not possible to estimate the actual medical-treatment costs and lost wages resulting from the use of a particular pesticide. Norton et al. (this volume Part III) identify and describe the four steps involved in estimating the impact of a change in pesticide exposure resulting from the adoption of an IPM practice. The first step is to identify the pesticide's risks to the environment and public health. Levitan et al. (1995), Kovach et al. (1992), Higley and Wintersteen (1992), and Mullen (1995) describe approaches used to rank pesticides by their degree of risk (e.g., low, medium, and high) in one or more vectors of concern. The second step is to quantify the effects of IPM adoption on the use and exposure to pesticides by their risk category. Developing an estimate of society's "willingness to pay" for reduced pesticide risk is the third step. Usually, the value to society of reducing that risk is not available. Contingent valuation (CV), a controversial but often employed technique, is an approach used to establish through opinion surveys monetary values for things not valued in the marketplace. When a CV approach is used, respondents are asked to make and value tradeoffs among environmental, public-health, and other reference goods (Mullen 1995, Higley and Winterstein 1992). This method derives estimates of society's "willingness to pay" for reductions in real or potential risk. The fourth and final step involves using these estimates to value the change in risk levels resulting from IPM practices. This monetary estimate of the public-health costs can then be incorporated into a comprehensive assessment of impacts.

Conclusions

IPM methods and technologies can have an impact on the entire ecosystem. Good IPM practices (such as inventory control, reduction of spill hazards, personnel training, pesticide formulation considerations, and product substitution) will reduce both worker and environmental exposures. The ability to demonstrate a reduced risk to humans from an IPM program should be a major selling point of such a plan. To accomplish this, one must know the health risks of the current practices and the potential risks from the new practices. Ongoing research efforts to evaluate the risk of cancer and other diseases among farmers, farm families, and farm workers from various agricultural chemical exposures will expand our knowledge about these critical relationships.

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Table 1. Causes of death showing deficits and excesses among farmers

Cause of Death	Number of Studies	Number with R/R* less than 1.0
Total mortality	10	9
Ischemic heart disease	12	12
All cancer	20	18
Lung	24	23
Bladder	21	19
Colon	15	13
Esophagus	18	12
Pancreas	20	11
Rectum	13	6
Kidney	15	9
Skin, nonmelanotic	8	4
Non-Hodgkin's lymphoma	14	5
Brain	18	5
Connective tissue	7	2
Prostate	22	6
Leukemia	23	9
Stomach	24	9
Multiple myeloma	12	2
Melanoma	11	2
Hodgkin's disease	12	2
Lip	8	0

*R/R = Relative risk

Table 2. Relative risk of non-Hodgkin's lymphoma and reported frequency of herbicide use among Kansas farmers using 2,4-D

	Number of Exposed Cases	Number of Exposed Controls	Relative Risk
Never farmed	37	286	1.0
Days per year of use			
1–2	6	17	2.7
3–5	4	16	1.6
6–0	4	16	1.9
11–20	4	9	3.0
21 or more	5	6	7.6

Table 3. Relative risk of non-Hodgkin's lymphoma and reported frequency of 2,4-D use among Nebraska farmers

	Number Exposed Cases	Number Exposed Controls	Relative Risk
Never farmed	54	184	1.0
Days per year of use			
1–5	16	44	1.2
6–20	12	25	1.6
21 or more	3	4	3.3

Table 4. Current and past use of protective practices among Iowa and North Carolina farmers

	10 Years Ago (%)	Currently (%)
Use rubber gloves		
Iowa	43	80
North Carolina	26	48
Use rubber boots		
Iowa	6	14
North Carolina	4	12
Change clothes immediately		
Iowa	5	9
North Carolina	20	30
Wash application clothes separately		
Iowa	63	81
North Carolina	50	68

A Primer on Economic Assessment of Integrated Pest Management

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Introduction

Scientists engaged in integrated-pest-management projects and programs are frequently asked about the benefits and costs of their IPM activities. They are asked to respond to such questions as:

- What is the impact of your IPM program?
- We spent $xx on your IPM program; what did we get for those funds?
- What are the environmental benefits of your IPM program?
- How profitable will IPM (or a particular IPM strategy) be for my farm?

Answering these questions requires practical assessment methods that are rigorous enough to provide credible responses yet cost-effective enough not to absorb too much of a total IPM budget. Using relatively standard evaluation methods can help ensure rigor and facilitate assessment of aggregate benefits across programs, but use of innovative assessment methods may also be required to evaluate difficult-to-measure impacts.

The questions posed above imply that the audience for impact assessments includes both (1) IPM users (e.g., farmers) interested in the benefits and costs of specific IPM tactics and strategies and (2) those responsible for funding and administering IPM projects and programs who are interested in more-aggregate impacts. Benefits can be measured at the level of the firm or for society as a whole. Goals for IPM include both economic profitability as well as environmental and health improvement. A range of methods are available to address these multiple dimensions of IPM impact assessment. Some of the methods require specialized training in economics while others do not. They all require adherence to certain standards for gathering and analyzing data if they are to provide believable results.

The purpose of this paper is to identify a core set of methods that can form part of virtually any IPM impact assessment and to highlight some of the possibilities for more complete analysis of IPM programs. Because defining IPM and measuring its adoption is a critical first step in any impact assessment, that topic is addressed first. Then, methods for basic economic assessment are presented, and finally, methods for environmental and health assessment are elaborated.

Defining IPM

A commonly understood, commodity- and location-specific definition of IPM is needed to define IPM and to measure its level of adoption. A process involving local stakeholders is recommended for establishing the definition, while recognizing that measures of IPM adoption will be used for impact assessment at various levels (local, state, regional, and national), and hence, some standardization in approach is needed to facilitate the more aggregate level assessments as well. The two aspects of standardization that can help in developing a definition that is workable across these levels are (1) agreeing on a common set of goals for IPM and (2) agreeing on a minimum set of levels into which the IPM continuum will be divided.

Goals

IPM can contribute to goals of (1) increasing income to IPM users and society as a whole through increased productivity and lower cost products and (2) enhancing environmental quality and health through reduced use of hazardous chemicals. These two primary goals can have several components as well. The process for establishing weights on these goals or their components should involve a broad spectrum of stakeholders.

Levels

IPM adoption is seldom a with-or-without situation because of the many potential practices involved and the fact that these practices are often adopted to varying degrees. Progress can be measured along vectors that express the extent to which progress has been achieved in meeting particular IPM goals through adopting individual or sets of IPM practices. In some studies, practices have been grouped to identify levels of adoption, such as none, low, medium, and high. In other studies, a continuous scale has been developed that gives points to different IPM practices. If scientists evaluating IPM programs could agree on using a scale with at least four levels, aggregation across programs would be facilitated. If a more detailed point scale were used, it could always be categorized down into these coarser levels if desired.

Process

The process of defining IPM can be flexible within each program, but should begin by defining the boundaries in time and space where the program is fairly homogeneous. Stakeholders for the IPM program must be identified, such as producers, scientists, extension agents, consumers, and others. Representatives of these stakeholder groups can be assembled and, with the help of a coordinator or facilitator, a participatory process can be used to identify existing IPM tactics or strategies that are available to control the pest problem(s) within the program boundaries. Once these tactics and strategies are identified, they can be grouped to delineate at least four levels of IPM adoption. The more data that can be supplied by scientists with respect to the effects of these IPM practices on production or pesticide use, the easier it will be to group them. Even with accurate data, the grouping will vary with the implicit weights attached by stakeholders to the income versus environmental goals.

This grouping of practices into levels of adoption on the IPM continuum is the most common method used for defining IPM adoption. It was used in the national IPM evaluation study in the mid-1980s (Rajotte et al. 1985), by the Economic Research Service, U.S. Department of Agriculture (Vandeman et al. 1994), and in a recent study by the World Wildlife Fund (Benbrook 1996) among others. An alternative to grouping practices is to attach points to the individual IPM tactics and strategies to derive a continuous scale. Stakeholders will vary the points they attach depending on their weights on economic versus environmental goals. An example of applying this point system procedure is provided by Hollingsworth et al. (1992) for the Massachusetts apple IPM program.

It makes little difference whether a set of levels or a continuous scale is used because either procedure can yield results amenable for project- or aggregate-level analysis. However, the makeup of the stakeholder group can influence the results because of the effect on weights applied to the two primary goals of IPM.

Basic Economic Assessment

A wide range of methods is available for assessing farm-level or more aggregate-level impacts of IPM on income, income risk, and the environment and health. These methods are seldom direct substitutes for each other, although often a particular method can be applied at different levels of detail. Also, the results of applying one method are frequently an input into a second method. For many difficult-to-measure impacts, particularly those related to the environment, additional research is needed to refine the methods, and many detailed IPM impact assessments are research projects in their own right. As a result, they can absorb significant time and resources. The intent in this section is first to highlight the various methods available for impact assessment and the resources required to implement them and then to discuss, in more detail, a core set of methods that can be used in virtually any basic economic evaluation of IPM.

Farm-Level Profitability

The primary method used for farm-level profitability analysis is to budget out the effects of changes in input and output quantities and prices as a result of adopting IPM practices. Budgets can be constructed as *enterprise* budgets, *partial* budgets, or *whole-farm* budgets. Examples of enterprise and partial budgets are provided below; but basically, enterprise budgets list all income and expenses (variable and fixed) associated with a particular enterprise, while partial budgets may include several enterprises but only include benefit and cost items expected to change significantly as a result of changes in production practices. A whole-farm budget includes all enterprises on a farm, and therefore can consider second-order changes in any activity as a result of introducing IPM practices. The most common types of budgets used for assessing IPM impacts are enterprise and partial budgets.

When budgeting is used to compare yields, costs, and profitability of IPM practices, statistical significance of differences should be tested. For example, if there are two groups of farmers, adopters and nonadopters, a t-test can be run to test for significant differences between mean yields, or analysis of variance can be used to test for significant differences among yields of a crop grown under three or four levels of IPM. However, it is generally preferable to test for significant differences in yields or profits with regression analysis with samples derived from populations of IPM adopters of different levels. For example, a yield-response equation can be estimated in which dummy variables are included to account for differences in IPM adoption. The t-statistics are then calculated for the coefficients on the dummy variables to account for significant differences, while other variables are included in the model to hold constant many of the non-IPM factors affecting yields. Masud et al. (1984) provide an example for delayed planting dates to control cotton bollweevils in the Texas Rolling Plains.

Results of budgeting analysis can be used by scientists and extension workers to judge the profitability of practices they are developing or will be recommending to farmers or of practices already adopted. A second major use of budget information is as an input into a more aggregate assessment of the economic benefits and costs of an IPM program as discussed below. The key audience in this case may be those responsible for funding the IPM program.

Farmers considering adopting particular IPM tactics or strategies are interested in their projected *profitability* as well as their economic *risk*. Risk may arise from biological, technical, or economic factors. A *payoff matrix* can be developed that lists projected net returns for different pest-management practices and severities of pests. (See table 1.)

Table 1. A hypothetical monetary payoff matrix for insect control per hectare

Pest Severity	Conventional	IPM
Light	$200	$350
Severe	$50	-$50

The decision to adopt a particular practice must be made before information is available on pest severity. Therefore, the decision will depend on the producer's ability to absorb risk and on an assessment of the probabilities of light or severe pest attacks. If historical information is available to help in calculating the probabilities, expected monetary outcomes could be calculated for each pest-management practice. In addition, the cells in the matrix could be subdivided to account for risks associated with crop prices and other factors. Pest forecasting can be used to provide information on the probability of a severe or light pest attack.

Additional discussion of payoff matrices is found in Reichelderfer, Carlson, and Norton (1984). An example of the use of economic analysis in a decision theory approach to crop-disease forecasting and control is provided by Carlson (1970).

The attractiveness of alternative pest-management practices to farmers in the presence of risk can also be assessed with a technique called *stochastic dominance* (SD). Stochastic dominance allows for comparisons of probability distributions to determine the most preferred choice for different classes of decision makers. There are three basic types of SD. First-degree SD ranks all distributions for all decision makers. Second-degree SD ranks distributions for risk averters. Unfortunately many distributions are left unranked with first- and second-degree SD. The third type of SD, called generalized SD, can be used to determine whether or not all producers in more narrow sets of risk preferences will prefer one cumulative distribution of net income associated with a management strategy or another or have no preference. Pairs of alternative pest-management strategies may be examined for various sets of producers. These sets of producers can be defined by their levels of risk aversion.

An example of the use of generalized SD in the economic evaluation and comparison of IPM strategies with conventional strategies for soybeans is found in Greene et al. (1985). Studies that use first- and second-degree SD include Musser et al. (1981), Moffit et al. (1983), and McGucklin (1983).

Farm-level economic evaluations of IPM programs are often concerned not only with the choice of practices but also with the optimal level of pest control with those practices. If profit maximization is assumed as the goal, optimal use of an IPM practice occurs when the marginal increase in net returns from applying another unit of the practice equals the marginal cost of its application. Entomologists in particular have applied this concept when identifying *economic thresholds* for pest densities. An economic threshold is the pest population that produces incremental damage equal to the cost of preventing that damage (Headley 1972). If the pest density is below this threshold, no treatment is justified. If it is above this level, treatment should occur to reduce pests to this level. IPM programs often involve monitoring or scouting to provide information to producers on pest densities in relation to the threshold.

The determination of what the economic threshold level should be is difficult because it is influenced by many factors. *Damage functions* are needed that relate pest levels to crop losses. Pesticide costs, output prices, effects of pesticide use on the development of pest resistance, and the effects on predators are other important factors that influence the threshold. And, if risk aversion on the part of producers and off-site costs of pesticide pollution are considered, economic thresholds might differ substantially from ones that only consider direct effects on net returns.

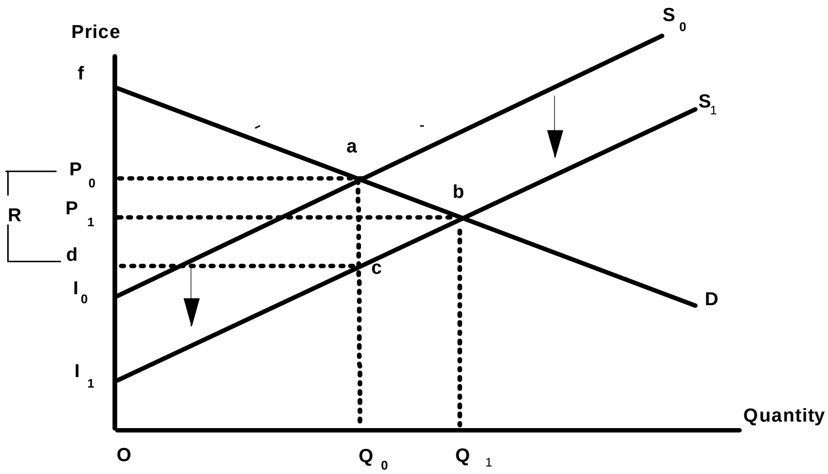
Several economists have studied optimal use of pest-management practices with *mathematical programming* techniques, such as *linear programming, nonlinear programming, and dynamic programming*. Linear programming maximizes an objective function (such as net returns from a set of cropping activities) subject to resource constraints (such as land, labor, capital, and water). Cropping activities can be included that incorporate various types of IPM practices. Enterprise budgets are incorporated in the model, and the sensitivity of the solution to changes in price and resource availability is easily examined. Linear programming assumes all activities and constraints can be cast in linear form. Martin et al. (1991) provides an example of an analysis of alternative tillage systems, crop rotations, and herbicide use on East-Central cornbelt farms. Nonlinear programming is an extension of linear programming that allows for nonlinear relationships. An application of nonlinear programming to a pest-management problem that includes pesticide resistance is found in Gutierrez et al. (1979). Dynamic programming allows for examination of optimal pest-control strategies when time is an independent argument in the models and the variables (such as plant product, pest population density, and the stock of pest susceptibility to pesticides) are all functions of time. Zacharias and Grube (1983) provide an example of applying such a model to examine optimal control of corn rootworm and soybean cyst nematode in Illinois.

Aggregate Economic Impacts

Methods for measuring the aggregate economic impacts of IPM programs on society as a whole can involve several techniques, but at the heart of these techniques is basic benefit-cost analysis. This analysis takes into account changes produced by IPM in production, costs, prices to producers and consumers, and the timing of these changes, giving greater weight to costs and benefits that occur sooner rather than later. Environmental and health effects can also be included if data are available. Methods for assessing environmental and health impacts are discussed in more detail below.

When widespread adoption of IPM occurs across large areas, changes in crop prices, cropping patterns, producer profits, and social welfare can occur. These differences arise because of changes in costs and because greater supplies affect prices to producers and consumers. These changes are illustrated in figure 1. In this model, S_0 represents

Figure 1. Research induces a shift down in the supply curve from S_0 to S_1 resulting in a cost saving per unit of output of R.



the supply curve before adoption of a set of IPM practices, and D represents the demand curve. The initial price and quantity are P_0 and Q_0 . Suppose adoption of IPM leads to a savings of R per unit in the average and marginal cost of production, reflected as a shift down in the supply curve to S_1. This supply shift leads to an increase in production and consumption to Q_1 (by $\Delta Q = Q_1 - Q_0$) and the market price falls to P_1 (by $\Delta P = P_1 - P_0$). Consumers are better off because they can consume more of the commodity at a lower price. Consumers benefit from the lower price by an amount equal to their cost-saving on the original quantity (Q_0 x ΔP) plus their net benefits from the increment to consumption. Although they may receive a lower price per unit, producers are better off, too, because their costs have fallen by R per unit, an amount greater than the fall in price. Producers gain the increase in profits on the original quantity (i.e., Q_0 x R – ΔP) plus the profits earned on the additional output. Total benefits are obtained as the sum of producer and consumer benefits. The distribution of benefits between pro-ducers and consumers depends on the size of the fall in price (ΔP) relative to the fall in costs (R) and on the nature of the supply shift. Examples of IPM evaluation that have assessed these income benefits to producers and consumers are found in Taylor and Lacewell (1977) and in Napit et al. (1988). Formulas for calculating consumer and producer gains and losses for a variety of market situations are found in Alston, Norton, and Pardey (1995).

Economists call this method of calculating economic gains and losses *economic-surplus* analysis. The most difficult component of an economic-surplus analysis is the calculation or prediction of the proportionate shift in supply following IPM adoption. Cost differences as well as adoption rates must be calculated or projected. Adoption rates are particularly difficult to estimate because they include changes in acreage as well as the proportion of producers adopting. Producer surveys can help in estimating adoption as discussed below. Several studies have estimated econometric relationships that assess factors influencing past adoption. These models can then be used to help predict future adoption. Napit et al. (1988), Harper et al. (1990), and Fernandez-Cornejo et al. (1992) provide examples in which *logit models* were used to estimate the relative importance of several socio-economic and other variables in influencing IPM adoption.

Once changes in economic surplus are calculated or projected over time, benefit/cost analysis can be completed in which *net present values*, *internal rates of return*, or *benefit/cost ratios* are calculated.

The benefit side is the total economic surplus calculated year by year, and the costs are the public expenditures on IPM programs. Benefit-cost analysis takes into account the fact that the sooner the benefits occur the more they are worth.

Changes in economic surplus can also be imbedded in mathematical programming models to predict interregional changes in production following the introduction of a widespread IPM program or to predict the impacts of IPM following policy changes that encourage and discourage IPM use. The interregional analysis can use *quadratic programming,* while policy models are likely to use dynamic programming (see, for example, Archibald 1984) or *dynamic simulation* (see, for example, Kazmierczak 1991). These dynamic models do not have standard algorithms and hence are more difficult to solve than the static (linear or quadratic) programming models. However, because the impact of IPM programs is inherently dynamic because of factors like pest resistance to pesticides, the results of dynamic models can be more realistic than static models if sufficient complexity is incorporated in themodels. The advantage of dynamic simulations over dynamic programming is the ability to add more complexity to an empirically tractable model.

Resources Required

The time, people, and financial resources required to implement the impact assessment methods highlighted above differ significantly. Enterprise or partial- enterprise budgeting, which are described in greater detail below, can be accomplished in a relatively short time (weeks) with little input needed from economists and with the primary costs involving surveys to identify cost differences by adoption levels. Likewise, simple payoff matrices can be constructed with little input from economists, although more complex risk analyses quickly become research projects in their own right, are greatly facilitated by input from economists, and can require several months to complete.

Most of the whole-farm-planning and mathematical-programming methods require the assistance of economists and several months time to complete. Likewise, the aggregate analyses involving economic surplus and benefit-cost analyses require collaboration between biological scientists and economists and can take several months. It is not the analysis itself that takes time, but the data collection.

Suggested Core Set of Methods

The suggested core set of methods for basic economic assessment of IPM include (1) a combination of enterprise and partial-enterprise budgeting and (2) benefit-cost analysis. The budgets can provide the field- and farm-level impact assessments required by producers, extension workers, and consultants for profitability assessments. They also generate information that is an input into the benefit-cost analysis required to demonstrate program impacts at a more aggregate level to those responsible for funding IPM programs.

Four basic steps in the economic assessment include:

1. Define IPM practices.
2. Define levels of IPM.
3. Identify production and input changes, and budget them out by adoption level.
4. Benefit-cost analysis to assess aggregate impacts.

Define IPM Practices. A participatory process as mentioned above with stakeholder groups including scientists, producers, consultants, and others can be used to identify the key pests and the tactics and strategies available to manage those pests within the program boundaries.

Define Levels of IPM. As discussed earlier, once the tactics and strategies are identified, the stakeholder groups should delineate at least four levels of IPM adoption: none, low, medium, and high. These levels will be based on subjective assessment of the contributions of the practices to

Table 2. Example of Baseline Survey Variables for Economic Analysis of IPM

1. *Inputs and Outputs (need quantity per acre, price per unit, percent acreage treated, number of times treated, method of treatment, who treated, etc).*
Herbicides, insecticides, nematicides, fungicides, labor for pest management, pheromone traps, scouting (self or hired), custom spraying, predators, outputs

2. *Extent of IPM adoption*
Practices used and percent of acres on which particular practices are used

3. *Pest problems and densities (in appropriate units)*
Arthropods, diseases, nematodes, rodents, birds, elephants, and weeds

4. *Producer and farm characteristics*
Farm size, acreage of crop, age and education of farmer, gender, years farming, ethnic identification, approximate value of farm, approximate value of farm products sold, and percent of income from farming

5. *Others*
Quality effects

economic and environmental goals. Each tactic or strategy can be listed on the board and then subjectively grouped based on these assessments. The assessments are inevitably subjective because unless one has already completed an economic and environmental assessment of the impacts of tactics and strategies, the stakeholder group can only provide very rough judgments on the contribution of the practices to each of the two goals. In other words, there is a bit of a chicken-and-egg problem in defining IPM. However, once some basic IPM impact assessment has been completed, future assessments are facilitated by the existing database.

Identify Production and Input Changes and Budget Them Out. Two primary options are available for gathering the necessary data to budget out the economic impacts of IPM. The first option is to conduct a baseline survey of producers in the area targeted by the IPM program with an interview. Questions should focus on (1) input and output quantities and prices that may change as a result of IPM, (2) pest problems and densities, (3) producer and farm characteristics, and (4) extent of IPM adoption. Basic-enterprise budgets available for the commodity and region are then modified based on the results of the baseline survey. Agricultural economists in the states involved and at the U.S. Department of Agriculture can help in locating the basic-enterprise budgets to be modified. A sample list of data needed is provided in table 2.

The second option is to construct complete-enterprise budgets from scratch by collecting information on all inputs by operation, preferably by having the farmers collect them in a standard tabular format as they do each operation, such as land preparation, planting, fertilization, pest management, cultivation, and harvesting. Data (quantities and prices) are collected on inputs like seeds, fertilizer, pesticides, labor, machinery use, and water and on all outputs. Pest population or pressure is measured as well. Data are also needed on output quantities and prices, quality (if relevant), and producer and farm characteristics.

Regardless of which of the two approaches is employed, a sample size of at least 30 per sample stratification group is required. For example, if pest management varies by farm size group (small, medium, and large), then the sample size should be at least 3 x 30 = 90. The costs of these two approaches can differ substantially, and the detailed collection of enterprise data by farm operation does not necessarily yield more accurate results if outputs and inputs vary substantially from year to year. A baseline interview survey can ask for estimated levels of the most important variables, say, for the past three years to help average out weather, pest, or price-induced differences across years.

Let us assume that the partial-enterprise budget data are collected through a baseline survey, rather than data for a complete-enterprise budget. Input and output quantities and prices are then entered into a budget form like the one shown in table 3. Total returns, costs, and net returns to management

Table 3. Enterprise Budget Form

	Unit	Price/ Unit	Low IPM Quantity	Value	Price/ Unit	Medium IPM Quantity	Value	Price/ Unit	High IPM Quantity	Value
Gross receipts	___	___	_____	_____	____	_____	_____	____	_____	____
Variable costs										
Preharvest (nonpest management)	___	___	_____	_____	____	_____	_____	____	_____	____
Preharvest (pest management)										
Insecticide	___	___	_____	_____	____	_____	_____	____	_____	____
Herbicide	___	___	_____	_____	____	_____	_____	____	_____	____
Nematicide	___	___	_____	_____	____	_____	_____	____	_____	____
Fungicide	___	___	_____	_____	____	_____	_____	____	_____	____
Scouting	___	___	_____	_____	____	_____	_____	____	_____	____
Labor and machinery	___	___	_____	_____	____	_____	_____	____	_____	____
Pheromone traps	___	___	_____	_____	____	_____	_____	____	_____	____
Predators	___	___	_____	_____	____	_____	_____	____	_____	____
Total preharvest costs	___	___	_____	_____	____	_____	_____	____	_____	____
Total harvest costs	___	___	_____	_____	____	_____	_____	____	_____	____
Interest on pest management variable costs	___	___	_____	_____	____	_____	_____	____	_____	____
Total variable costs	___	___	_____	_____	____	_____	_____	____	_____	____
Total fixed costs	___	___	_____	_____	____	_____	_____	____	_____	____
Total costs	___	___	_____	_____	____	_____	_____	____	_____	____
Return to Management	___	___	_____	_____	____	_____	_____	____	_____	____

are then calculated for IPM adopters at different levels of adoption. These results can be presented to producers by IPM extension workers and private consultants to demonstrate the profitability of IPM adoption. The results can also be incorporated in an aggregate benefit-cost assessment of IPM programs and shown to those who administer or fund the programs.

Benefit-Cost Analysis to Assess Aggregate Impacts. Aggregate-impact analysis takes the differences in costs per unit of production for different levels of IPM adoption; combines them with information on the geographical spread and timing of adoption; and (1) projects the economic benefits year by year produced by previous and/or potential IPM adoption, (2) discounts the annual benefits to account for the fact that benefits received sooner are worth more than benefits received later, and (3) compares the discounted benefits to discounted costs of the IPM program to produce a net present value or benefit-cost ratio. A rate of return on the IPM investment can also be calculated.

Benefit estimates can be generated by comparing cost differences across IPM levels with information from the baseline survey, estimating the length of time that these practices have been used, and projecting continued IPM adoption in the future. Alternatively, the baseline results can be compared with the results of a followup survey administered in a future year. A third alternative is to gather the information on the baseline survey and then to project the extent and timing of adoption with estimates by stakeholder groups.

Let us assume that the benefit of $20 per acre is estimated for use of high as compared to a medium level of IPM based on a baseline survey and partial-enterprise budgeting. Let us assume that it is estimated that 20,000 acres will be under the high level of IPM and that the acreage will be maintained for the next 10 years. Furthermore, let us assume that the program that produced the IPM practices took 5 years, cost $100,000 per year, and was completed last year. If we assume a discount rate of 5 percent and no price effect caused by the additional production that might result from the lower cost of production, an economist would then calculate the net economic benefits as the discounted benefits less the compounded costs:

$$= \sum_{t=0}^{4} \frac{400{,}000}{(1.05)^t} - \sum_{t=1}^{5} (100{,}000)(1.05)^t$$

$$= 1{,}818{,}380 - 580{,}191$$

$$= 1{,}238{,}189 .$$

If the influence of the IPM program was such that the lower cost and resulting production increase were large enough to influence the price of the commodity, economists would model the market as well, use a formula to estimate the economic benefits from a graph such as figure 1, and estimate or project the benefits for each year. For example, the benefits are equal to the area I_0abI_1 for a market situation with no trade, such as the one illustrated in figure 1. The formula to calculate these benefits is $KP_0Q_0(1 + 0.5Zn)$, where:

K = proportionate cost change
P_0 = initial price
Q_0 = initial quantity
Z = Ke/(e+n)
e = supply elasticity
n = demand elasticity

Other formulas would be appropriate for other market situations. Although the formulas presented in this section are not complex, biological scientists would be well advised to involve economists in this type of aggregate-impact assessment.

Methods for Environmental and Health Assessment

Increased attention has focused in recent years on the actual or potential environmental benefits of IPM. Measurement of these benefits is difficult for two primary reasons. First, assessing the physical or biological effects of alternative levels of pesticide use under different IPM practices is challenging. Second, the economic value associated with environmental effects is generally not priced in the market. The first problem has been addressed in studies by Kovach et al. (1992), Higley and Wintersteen (1992), and Mullen (1995). Kovach et al. divided the environmental effects into farmer, worker, consumer, and ecological components and used a variety of databases on the toxicity of pesticides in different settings to classify and weight the environmental impacts of pesticides, based on dermal toxicity, chronic toxicity, systemicity, fish toxicity, leaching potential, surface-loss potential, bird half-life, soil half-life, bee toxicity, beneficial arthropod toxicity, and plant-surface half-life. This weighting allowed them to arrive at an *environmental-impact quotient* by pesticide. They then multiplied this quotient by the percent active ingredient and application rates to obtain an environmental rating for the pesticide in field use. They compared the environmental impacts of traditional and IPM strategies; but they did not attempt to place an economic value on the differences in environmental impacts.

Higley and Wintersteen assessed the environmental risks of pesticides on three broad areas of environmental risk (water quality, nontarget organisms, and human health) that were then subdivided into eight specific categories [surface water, groundwater, aquatic organisms, birds, mammals, beneficial insects, humans (acute toxicity), and humans (chronic toxicity)]. They then classified each pesticide into high risk, medium risk, low risk, or no risk for each environmental category based on a set of criteria from several different studies. Mullen used a similar set of environmental categories.

Unlike Kovach et al., however, Higley and Wintersteen as well as Mullen tackled the issue of placing a value on benefits not priced in the market. They each used *contingent valuation* (CV) to assess the relative importance that individuals place on the environmental-risk categories and the amount they would be willing to pay to avoid high, moderate, and low levels of risk from a pesticide application. Higley and Wintersteen surveyed 8,000 midwestern producers. They used the results to estimate the environmental costs per pesticide. Mullen surveyed 3,000 households throughout the United States. He went a step further and estimated the effects of IPM adoption in apples and peanuts in Virginia on pesticide use. He then used the results of the CV analysis to calculate the economic value of the environmental benefits of IPM.

Contingent valuation is one of the few procedures available for estimating environmental costs associated with pesticide use (or environmental benefits of IPM if pesticide use declines). The procedure has been used for roughly 20 years (and particularly in the past 10 years) in other settings to estimate nonmarket costs or benefits. Typically, CV studies provide respondents with information about a hypothetical action that would reduce the likelihood of a future environmental problem, such as pesticide exposure to fish. Respondents are given some specific information about the nature of the damages. They are then confronted with a question or questions about the maximum amount they would be willing to pay to reduce the problem.

The CV technique has been controversial. Some have argued that respondents give answers that are irrational, that they do not understand what they are being asked to value, and that they do not take the questions seriously because they are hypothetical (Arrow et al. 1993). Others have argued that these problems can be minimized with carefully designed and administered surveys. Arrow et al. provide a detailed discussion of these issues.

The CV technique is one of the few procedures currently available for estimating the aggregate environmental benefits of IPM programs. However, other methods could be used for specific types of environmental effects. For example, hospital records on the costs associated with acute pesticide poisonings, insurance costs for farmworkers exposed to pesticides, costs of restoring polluted wildlife habitats, and other partial market-based techniques can be used in some situations. Antle and Pingali (1994) and Rola and Pingali (1993) have assessed the economic value of acute human-health effects associated with pesticide use in the Philippines. They considered both the medical costs and the effects of health problems on farmworker productivity. A related study was completed by Chrissman and Antle in Ecuador.

The whole area of valuing environmental benefits of IPM is flush with possibilities for close collaborations between biological scientists and economists. Biological scientists can continue to refine our knowledge of the physical or biological effects of pesticide use on various aspects of the environment and health. Economists can continue to refine methods for valuing these effects.

At the moment, it appears that CV analysis may be the one method available that can be used to place a value on the range of environmental and health effects of IPM in a cost-effective manner. Therefore, the section that follows describes the steps in implementing such an analysis.

Steps in a Basic Environmental Assessment of an IPM Program

Assuming that the level of IPM adoption has already been defined for a particular crop and region as discussed above, four basic steps are required for environmental and health assessment of an IPM program:

1. identifying pesticide risks to the environment,
2. assessing the effects of IPM adoption on pesticide use,
3. estimating society's willingness to pay for reduced pesticide risks, and
4. calculating reduction in risk levels and ap-plying willingness-to-pay estimates to them.

These steps were applied in an analysis of the environmental benefits of the apple and peanut IPM programs in Virginia by Mullen (1995) and are summarized in a paper by Mullen et al. (1996). The following is a brief summary of the steps with results presented for the Virginia peanut IPM program.

Identifying Pesticide Risks to the Environment

Pesticide risk to the environment is related to the amount of active ingredients (a.i.) applied. However, total pounds of a.i. applied per year is not the best measure of risk because pesticides differ with respect to their toxicity, mobility, and persistence. A given pesticide also may pose different levels of risk to different components of the environment. Substitution of one pesticide for another may reduce the risk to one component but raise it to others. To address this issue, the environment can be divided into eight broad categories (groundwater, surface water, acute human health, chronic human health, aquatic species, birds, mammals, and arthropods) and three levels of pesticide risk can be identified (high, moderate, and low).

Active ingredients can be assigned one risk level (j = 1 to 3) for each environmental category (i = 1 to 8), resulting in 24 risk/environmental classes for pesticides. Rather than measuring the change in total pounds of all a.i., it is preferable to measure the change in pounds of a.i. in each *ij* pesticide class attributable to IPM adoption. Separate criteria can be used for each environmental category to classify the risk posed by each a.i. The following is a brief summary of how risk levels were assigned to each a.i. for each environmental category in Mullen et al.

The assignment of groundwater risk to an active ingredient was based on the Pesticide Leaching Matrix developed by the U.S. Department of Agriculture Soil Conservation Service (USDA/SCS) (Becker et al.). The matrix accounts for both soil and pesticide leaching properties. If a pesticide-leaching rating was not available, Gustafson's Ubiquity Score was used to assign groundwater risk to the pesticide. Likewise, the assignment of surface water risk to an a.i. was based on the Surface Runoff Matrix developed by USDA/SCS. If a surface loss rating was not assigned to a pesticide, Red Flag values for water solubility, soil K_{OC}, and soil half-life developed by the EPA were used.

The assignment of acute human-health risks was based on signal words assigned by EPA to the formulated product. EPA requires all pesticides to be labeled with Danger, Warning, or Caution, depending on toxicity LD_{50}s for oral, dermal, and inhalation exposure; and eye and skin effects. (LD_{50} is the dose that kills 50 percent of the test population.) Criteria for assigning chronic-health-risk levels were based on the results of tests evaluating teratogenicity, mutagenicity, and carcinogenicity of each pesticide.

Aquatic species' risk levels were based on LD_{50}s and a weight for surface-water risk (because a pesticide cannot pose a risk to aquatic species if it does not reach surface waters). Assignment of risk of a pesticide to avian and mammalian categories was based on LC_{50}s and the highest level of risk to any species within the category. To assess risk to nontarget arthropods, several references were consulted, including EXTOXNET; Smith, Higley and Wintersteen; Kovach et al.; Worthington, Hartley, and Kidd; and EPA reregistration reports.

Assessing Effects of IPM on Pesticide Use

To estimate the reductions in external costs attributable to an IPM program, an estimate is needed of the proportional change in pesticide use induced by adoption of IPM on the study crop. Estimating this change entails comparing the current level of use under IPM to an estimate of what use would be in the absence of the IPM program.

Total pounds of an a.i. class applied per year to a study area can be denoted Use_{ij}, where i = environmental category and j = risk level as defined above. Use_{ij} is composed of two elements, use on the study crop (Use_{ijs}) and use on other crops in the study area (Use_{ija}) so that

$$Use_{ij} = \sum_{a=1}^{n-1} (Use_{ija}) + Use_{ijs},$$

where *n* is the number of crops grown in the study area.

Regression analysis can be used to examine the relationship between Use_{ijs} and various levels of adoption of IPM. A general form of this relationship can be represented by:

$$Use_{ijs} = F(\textit{IPM adoption, acreage of the study crop, pest severity, farmer characteristics})$$

For example, the four levels of IPM adoption defined above can be included as dummy variables and variables such as farm size, age, farmer education, and an index of pest infestation severity can be included. Realized and potential proportional reductions in Use_{ij} can then be calculated by comparing Use_{ij} with and without IPM.

Willingness to Pay to Reduce Pesticide Risks

Estimates are needed of society's willingness to pay to avoid pesticide risks to the eight environmental categories. There are few market proxies for the value of avoiding risk to any of these categories and none that would serve for all of them. Therefore, Mullen administered a contingent valuation survey (CVS) to a random sample of 3,000 U.S. residents.

The survey contained an introduction with a brief overview of the value of pesticides as an agricultural input and of the potential for pesticides to damage the environment and human health. The questionnaire began by asking the respondent's average monthly grocery bill. This question was relatively easy to answer and served to get the respondent involved in the survey. It also provided a baseline for a subsequent question on willingness to pay.

The willingness-to-pay (WTP) questions began with a brief definition of "high risks to the environment and human health from pesticide use." Respondents were asked their willingness to pay to avoid high risks via an increase in their monthly grocery bill. This payment vehicle was chosen because grocery prices might increase if the use of an entire class of pesticides was restricted. After answering the WTP questions, the respondents were asked to rate (from 0 to 6) how important it is to avoid high risks to each of the eight environmental and human-health categories considered in the study. The same format (risk definition, willingness-to-pay questions, and assignment of importance levels) was repeated for moderate and low risks.

The survey was mailed to individuals drawn randomly from motor vehicle registration records and telephone directories throughout the United States. A second mailing was sent 25 days later to 833 addresses, selected at random from those that had not returned the survey. Several surveys (384) were returned as undeliverable, and 454 responses were received.

To minimize the length of the questionnaire, the CVS respondents were asked to reveal their willingness to pay to avoid a given level of risk to the environment as a whole (WTP_j), rather than their willingness to pay for each category (WTP_{ij}). The importance rankings by category from the survey were then used to infer the respondent's WTP_{ij} from their WTP_j.

$$WTP_{ij} = \frac{importance_i}{\sum_{i=1}^{8} importance_i} \times WTP_j$$

The results of the CVS, with 46 outliers deleted, are presented in table 4. Following previous studies (Desvousges et al. 1993), responses were considered outliers if the WTP_j exceeded 5 percent of the respondent's annual income.

Calculating Risk Reductions and Applying Willingness-to-Pay Estimates

Risk reductions produced by reduced pesticide use resulting from IPM adoption can be combined with the willingness-to-pay estimates to assess the economic value of environmental benefit of IPM. The following is an example of such an analysis for peanuts in Virginia.

Table 4. Willingness to Pay to Reduce Environmental Risk ($/month)

Environmental Category	High Risk Mean	High Risk Std Dev	High Risk N	Moderate Risk Mean	Moderate Risk Std Dev	Moderate Risk N	Low Risk Mean	Low Risk Std Dev	Low Risk N
Acute Human	4.28	4.68	397	2.89	3.44	392	1.74	2.75	388
Chronic Human	4.59	4.85	397	3.14	3.68	392	1.89	2.87	388
Groundwater	4.56	4.75	397	3.08	3.62	392	1.86	2.91	388
Surface Water	4.40	4.62	397	2.93	3.43	392	1.76	2.79	388
Aquatic Species	4.37	4.64	397	2.88	3.42	392	1.75	2.84	388
Avian Species	4.15	4.48	397	2.72	3.23	392	1.63	2.67	388
Mammalian Species	4.13	4.46	397	2.71	3.25	392	1.65	2.69	388
Arthropods	3.76	4.33	397	2.49	3.11	392	1.50	2.54	388

The Virginia IPM program in peanuts focused on developing a disease-forecasting system to reduce fungicide use. In 1979, an early leaf spot advisory system (ELSA) was developed in Virginia to identify environmental conditions favorable to early leaf spot infection. Prior to ELSA, the conventional method for combating early leaf spot in Virginia peanuts was to apply chlorothalonil to peanut fields at 14-day intervals. By accurately predicting periods of early leaf spot infection, the ELSA forecasts and fungicide recommendations have allowed farmers to apply chlorothalonil in a more judicious manner.

In a four-year evaluation study from 1987 to 1990, it was found that farmers following ELSA recommendations made, on average, 33 percent fewer applications of chlorothalonil than farmers using the 14-day spray regime. Yields from the ELSA farms were not significantly different than yields from the 14-day spray farms; nor was there a significant difference in the value of those yields. By 1990, 94 percent of Virginia's peanut producers were applying chlorothalonil based on ELSA recommendations (Phipps 1993).

Recall that Use_{ij} is comprised of two components, the total amount of a.i. class *ij* applied to all crops in the study area other than the study crop (Σ Use_{ija}), and the total amount of a.i. class *ij* applied to the study crop (Use_{ijs}). The calculation of Use_{ija} is represented by

$$Use_{ija} = \sum_{p=1}^{m} (Acres_a \ x \ Treat_{ap} \ x \ Rate_{ap})$$

where *m* = number of active ingredients of class *ij* applied to crop *a*, $Acres_a$ = number of acres of crop *a* harvested in the study area, $Treat_{ap}$ = proportion of study area acres of crop *a* treated with active ingredient *p*, and $Rate_{ap}$ = pounds of active ingredient *p* applied per acre per year to crop *a*.

Similarly, $Use_{ijs,w/ELSA}$, the amount of active ingredient of class *ij* applied to peanuts in the study area in 1992, is calculated by

$$Use_{ijs,w/ELSA} = \sum_{p=1}^{m} (Acres_s \ x \ Treat_{sp} \ x \ Rate_{sp})$$

where *m* = number of active ingredients of class *ij* applied to peanuts, $Acres_s$ = number of harvested acres of peanuts in the study area, $Treat_{sp}$ = proportion of study area peanut acres treated with active ingredient *p*, and $Rate_{sp}$ = pounds of active ingredient *p* applied per acre per year to peanuts.

The total amount of a.i. class *ij* applied to all crops in the study area in 1992 is given by

$$Use_{ij,w/ELSA} = \sum_{a=1}^{n-1} (Use_{ija}) + Use_{ijs,w/ELSA}$$

where *n* is the number of crops grown in the study area.

Assuming that producers following ELSA recommendations applied 33 percent less chlorothalonil in 1992 than producers using a calendar spray schedule *and* that 94 percent of Virginia's peanut producers used ELSA while 6 percent used calendar sprays, one can solve for the amount of chlorothalonil that would have been applied in the absence of ELSA with the equations

$$X = 1.5 \text{ x } Y \quad \text{and}$$

$$Z = \text{Acres}_s \text{ x } (0.94 \text{ x } Y + .06 \text{ x } X) \text{ ,}$$

where X is the pounds of chlorothalonil applied per acre per year to farms with a 14-day spray schedule, Y is the pounds of chlorothalonil applied per acre per year to farms following ELSA recommendations, Acres_s is the number of peanut acres harvested in the study area in 1992, and Z is the total pounds of chlorothalonil applied to peanuts in the study area in 1992.

The amount of a.i. class *ij* that would have been applied to the study area without ELSA, $\text{Use}_{\text{ij,w/o ELSA}}$ is calculated as

$$Use_{ij,w/oELSA} = \sum_{a=1}^{n-1} (Use_{ija}) + \sum_{p=1}^{m-1} (Use_{ijsp}) + X^{*} \; Acres_s$$

where n is the number of crops grown in the study area, p is the number of active ingredients of class *ij* other than chlorothalonil applied to peanuts in the study area, and X x Acres_s is the total pounds of chlorothalonil that would have been applied to the study area in the absence of ELSA. The estimates of $\text{Use}_{\text{ij,w/ELSA}}$ and $\text{Use}_{\text{ij,w/o ELSA}}$ for the relevant a.i. classes are presented in table 5.

The savings in the external costs inflicted on each of the environmental-risk categories are represented by: $Savings_{ij} = WTP_{ij}$ x POP x $Realized_{ij}$, where POP is the population in the study area and Realized_{ij} is the realized proportionate reduction in Use_{ij}. The total savings in external costs (environmental benefits) attributable to the ELSA program is simply the sum of the savings for each of the eight relevant *ij* categories (table 5). The total savings in external costs are approximately $844,000 per year (in 1992 dollars).

The willingness-to-pay estimates developed in the Mullen study can be applied in other studies without the need to repeat the CVS. Procedures developed for assessing risk levels to eight environmental categories can also be used elsewhere. Risk levels were assigned to more than 130 pesticidal active ingredients in Virginia, and some of these results should be useful in other studies as well. Tables with these risk levels are available from the authors, and their availability can reduce the time and effort required in future studies. These risk assignments may also be used by farmers to guide their selection of pesticides.

Conclusions

A variety of approaches are available to assess economic and environmental impacts of IPM programs. Most of the approaches require collaboration between biological scientists and economists. It is possible to complete partial-enterprise budgets with relatively little assistance from economists. However, most aggregate-impact assessments aimed at audiences like administrators or funding agencies require a multidisciplinary approach in which, at a minimum, economic-surplus and benefit-cost analyses are completed. Some progress has been made in assessing the economic value of environmental benefits, but this topic is ripe for additional research. If pesticide reductions from IPM are estimated as well as hazard levels of those pesticides, the willingness-to-pay estimates provided in table 4 can be used to assess the economic value of the environmental benefits of an IPM program.

Table 5. Estimates of Chlorothalonil Use With and Without ELSA and Savings in External Costs (Environmental Benefits)

Active Ingredient Class	$Use_{ij,w/ELSA}$ (1000 lbs)	$Use_{ij,w/o\ ELSA}$ (1000 lbs)	Percent Reduction in Use_{ij} Produced by ELSA	Savings in External Costs (1000 $)
Low risk to groundwater	747	844	11.56	142
High risk to surface water	1937	2035	4.80	139
High risk to aquatic species	1857	1954	4.99	144
High risk to acute human health	1745	1842	5.30	149
Moderate risk to chronic human health	2268	2366	4.13	85
Low risk to avian species	2241	2338	4.17	45
Low risk to mammalian species	965	1063	9.18	100
Low risk to nontarget arthropods	2325	2423	4.03	40
Total				844

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Practical Considerations in Assessing Barriers to IPM Adoption

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The charge to the rural sociologists participating in this session was simple and direct: "no theory, no research findings, just practical explanations of what your discipline has to offer to those promoting the adoption of integrated-pest-management (IPM) practices." Asking scientists to make presentations without theory or data was difficult, yet achievable, when considering the objectives of this session. The challenge facing the rural sociologists in this session was to find a balance between providing a one-size-fits-all "cookbook" of IPM adoption on the one hand, and losing the audience with myopic research detail on the other. Instead, the presenters were asked to provide practical recommendations on how social processes could be applied to increasing IPM adoption.

In the first section of our paper, we raise a number of important issues regarding the foundation upon which higher levels of IPM adoption are expected to occur. A critical question is associated with the value placed on information, the very substructure upon which IPM recommendations are developed. That is, producers engaged in integrated pest management collect, analyze, and use information as the basis for pest-management decisions. This requirement for the analytical use of quality information occurs in a context where the producer is often overwhelmed by diverse data sets (e.g., markets, weather, new technologies, input prices, farm programs, and community activities). As we point out, it is into this context that IPM programs are trying to get producers to recognize and use quality information. We develop the argument that current adoption levels may represent the "easy" cases, and either enhancing the level of adoption or persuading remaining nonadopters to attempt IPM practices may require qualitatively different initiatives. It cannot be "more of the same" if we are to achieve the 75-percent adoption objectives.

In the second section, we discus IPM adoption and barriers to this process. As is the case with most complex phenomena, measurement is a critical issue. Adoption of IPM practices can be measured on different levels. We describe four levels of measurement associated with IPM practices. These levels more or less represent a continuum from simple measures that characterized past program-accountability efforts (accounting level of measurement), current efforts (proportional level of measurement), future efforts based on site-specific accuracy in using IPM practices, to the distribution of those practices across an ecological landscape. We identify and discuss a set of barriers producers encounter when faced with IPM-adoption decisions. This analysis is based on understanding IPM adoption from the perspective of the producer. This contribution concludes by noting that producers are making correct and rational decisions in rejecting IPM recommendations because of the presence of one or more of these barriers. Those interested in increasing IPM adoption rates are encouraged to address these barriers rather than blaming the farmer for current nonadoption decisions.

In the third section of this paper, we examine some of the social processes that often impact IPM program efforts. Successful IPM programs are usually based around the cooperative efforts of multiple agencies, organizations, firms, and producer groups. These partnerships do not "just happen," but require careful planning and support. We discuss some of the factors associated with conflict management, building consensus, and improving communication critical to the success of these partnerships. Finally, we address the difficult issue of social-impact assessment associated with IPM adoption. The diffusion of IPM across a production region or commodity will produce "winners and losers" as a consequence of that process. IPM program managers and professionals need to be aware that their efforts will have these impacts. How to assess and manage these impacts are a final theme in our presentation.

Overcoming the Plateau in Adoption of Integrated Pest Management

The benchmark of 75 percent of the nation's managed acres under IPM by the year 2000 is a challenging but justifiable goal. After all, IPM has been promoted and publicly funded for more than a generation. On the optimistic side, the goal might just be achievable. If a less rigorous definition of IPM is invoked and self-report data are used, then current levels of IPM adoption would be regarded as relatively high and within reach of the benchmark (Vandeman et al. 1994). For insect control in corn, we might already be at the benchmark. For weed control in corn, adoption may be about two-thirds of the benchmark, and perhaps adoption is as high as 80 percent for weed control in soybeans (Vandeman et al. 1994). However, if more stringent definitions are used, such as those proposed by some interest groups, the level of adoption may be regarded as half or less of these levels (Cate and Hinkle 1994).

The Foundation for IPM

A key question is, "Has the foundation been laid with producers for doing the right things for the right reasons?" Fundamentally, IPM is a "whole-system," intensive, and information-based management approach. As such, IPM cannot be reduced to a cafeteria of independent or substitutable practices. Unfortunately, much adoption literature is based on single-practice innovations, and too frequently analysis has been approached from a simple-technology perspective. Understandably, this is a consequence of subject-matter specialization and the setting of parameters for scientific inquiry. Unfortunately, solutions based on substituting one technology for another have limitations when extrapolating to an integrated-systems approach, where there may be many acceptable (desirable) solutions to the puzzle. This could include some solutions that, when taken at face value or in a single time frame, may appear contradictory to the overall intent of the system being advocated (i.e., unique incidents where high levels of pesticide use is warranted). Or, conversely, a set of practices might appear to be consistent with IPM but are not rooted in the systems approach underlying most concepts of IPM.

IPM as a Process

Without question, defining IPM is an ongoing process, both in the abstract and in practice. Debate continues regarding the importance of certain goals and priorities for IPM, such as use of and dependence on chemical pest-control practices (Gray 1995). Nonetheless, there is general agreement that IPM is an information-based approach providing multiple options for pest control based on sound data inputs. Underlying all of this, then, is an essential ingredient, namely an information base generated from on-farm or site-specific observations. Meaningful pest scouting and subsequent documentation from the scouting activity must be central to decisions if a management system is to qualify as IPM. What is less clear is the extent to which crop producers value and appreciate the importance of such site-specific information in pest control and overall crop production and whether producers have identified with the systems approach it represents. The extent that this foundation is not a motivational factor represents a major barrier to full IPM adoption. IPM is a managment and information-intensive pest-management system and should be acknowledged as a version of "precision farming," even though it has neither the glamour of nor dependence upon Geographical Information Systems (GIS) or Global Positioning Systems (GPS) technologies. Broadening the definition to include IPM would allow producers to garner some of the benefits of precision farming without adopting these new and developing technologies.

IPM in the Information Age

Only when the value of on-site data is well understood and incorporated in the decision-making process has the foundation for IPM been established. Therefore, when attempting to "sell" IPM, it needs to be done from an integrated-systems and information-age perspective and not as a list of individual practices. Further, its advocacy must be undergirded with the values, norms, and technologies (i.e., the culture) of the information age. This involves at least a few important departures from the mass production, mass society, and economy-of-scale agriculture framework that has been and remains pervasive among producers.

The information age should not be mistaken as merely more information or intrigue associated with information-age technologies, such as the computer chip, the Internet, home pages, GIS, GPS, or variable-rate technology (VRT). The promise of the information age is that information would be different from mass society/mass media information. The notion that one size (with a little alteration) will fit most, if not all, situations, is replaced with the expectation that information must be custom-designed for each site-specific situation.

As producers perceive the beneftis from site-specific information, it begins to have a market value. That makes information a commodity or product akin to other inputs into the production process. In the case of IPM, pest scouting is an example of a type of information input. In the case of integrated crop management (ICM), additional kinds of site-specific information (nutrient levels, yields, soil types, crop rotation histories, etc.) are part of a more complex mix. And the value of this onsite-generated data is realized when it is interpreted and juxtaposed against more generalized research-based findings and principles. In short, production information and pest-control information must be more than dealer sales counter calculations that use a general formula and a few rough data estimates.

Producing, recording, analyzing, and applying site-specific data in conjunction with more general research-based knowledge does not come without a cost, either in the form of a purchased service or a direct investment in time and effort by the producer. Failure to recognize the importance to invest in quality information may well put a ceiling on full adoption of IPM and thereby limit the production and environmental benefits that potentially can accrue from more universal adoption of IPM. While the data are somewhat ambiguous, a recent study of corn and soybean producers in Iowa point to a reticence to identify with and commit to the importance of quality onsite and farm-produced data in decision making.

IPM and On-Farm Data

Findings from several surveys of corn and soybean producers in the Midwest provide insights into current production practices and suggest the need to more strongly reinforce the value of onfarm-produced data as the basis for promoting increased adoption of IPM. Sample surveys conducted in Iowa suggest that farmers in that state do not universally identify with IPM. Indeed, less than 10 percent of farmers say they make "heavy use" of IPM, with perhaps as many as one-quarter who identify with "moderate" or "heavy" use. Fully two in five say they do not use IPM. This percentage has been fairly stable for the past five years (Lasley 1989; 1994). When asked about individual cultural practices to limit dependence upon pesticides, certain practices are adopted much more widely than IPM itself (mechanical cultivation), while others (banding herbicides and using degree days) are quite similar to IPM in extent of use (Lasley 1994). In the past several years, information providers in the state [Extension, National Resources Conservation Service (NRCS), and several in the private sector] have been promoting the broader concept of ICM, which also makes strong application of onfarm and site-specific information. At least for now, identification with ICM among Iowa's corn and soybean producers is less than for IPM (Lasley 1994).

On the critical issue of scouting, the findings look quite promising on the surface, but in-depth questioning elicits concern about whether a number of farmers have deceived themselves on what constitutes acceptable and rigorous scouting for high-management IPM needs. Again, the Iowa Farm and Rural Life Poll has quizzed Iowa farmers on scouting, and this has been augmented in other surveys as well. Most (90 percent) Iowa farmers indicated they make at least limited use of scouting and one in five indicate "heavy" use (Lasley 1994). However, when Iowa farmers were asked the question in a slightly different way, namely how many times do they walk their fields to specifically check for the presence of insects, weeds, diseases, or other problems, half of the farmer respondents indicated three times or less per growing season, and less than one-fifth indicated a half dozen times or more (Padgett 1990). Although Czapar et al. (1995)

found higher levels of farmer scouting among central Illinois farmers, their sample may have included more large-scale grain farmers than the Iowa surveys.

Particularly the Iowa studies, but to some extent also the Czapar et al. Illinois study, raise questions about whether farmers see the importance of rigor in recording, using, and incorporating scouting information for management purposes. Currently, most scouting is done by the farmers themselves, and professional crop scouting is relatively infrequent. Both Lasley and Czapar et al. report use of professional crop scouting by approximately 7 percent of their study respondents. General crop consulting may be at a higher level, however. An extrapolated estimate by Doane Agricultural Services (1993) places professional crop consulting nationally at 21 percent for corn and at 12 percent for soybeans. Padgett (1990) found that cost was a major factor inhibiting Iowa farmers from purchasing the servies of professional scouts or crop consultants. At the time his study was conducted, approximately 5 percent of the farmer respondents indicated an interest in professional scouting when priced at the market rate, but as many as one-third expressed an interest if the scouting was offered at about one-half the existing market rate.

When a pilot effort, the Model Farms Project, was launched in Iowa, it provided incentives for integrated crop-management services, including systematic scouting. Interest was high in the initial identification of project cooperators. But, by the time user fees were incrementally increased over a three-year period to a competitive market level, approximately one-half of the original cooperators left the program (Petrzelka, Padgitt, and Wintersteen 1995). The most frequent reason cited among those leaving the program was that they did not see sufficient economic benefit from the crop consulting services. Nearly two-thirds of those leaving the program (64 percent) gave this as a reason, with just slightly fewer noting they could not financially afford to continue the service (58 percent). These reasons are in sharp contrast and surpass the frequency with which they noted yield loss (4 percent), incompatibility with their current production system (8 percent), and inability to control weeds and insects (11 percent). Also, the lack of economic benefit contrasts sharply with project records, which document the return on investment to be more in the range of four to one at full market value. The value is reasonably close to estimates made by continuing clients of Iowa's private crop consultant, who in a 1993 survey reported by a margin of four to one that their benefit exceeded the investment, and nearly half of whom said the rate of return was at least quadruple the investment (Petrzelka et al. 1995).

Finally, among the profile of Model Farms cooperators in Iowa (farmers selected because they were more forward looking than the average producer), many did not keep and use field-based records, the kinds of records that facilitate information-intensive management decisions. However, over the course of the project, most of those who remained in the project did change and adopted to a much greater extent the notion of site-specific record keeping. Their changes were substantial and document progress because record keeping is not a highly enjoyed activity by Iowa farmers, especially when compared to crop and field work (Lasley 1992). Consequently, Iowa farmers reported spending very little time at it. Sixty-two percent of respondents to the 1992 Iowa Farm and Rural Life Poll reported investing 5 hours or less per month keeping and analyzing records. This is not the profile needed in a management-intensive, information-age production system and lends credence to the notion that, while adoption of individual IPM practices may be increasing, the decisions are likely based on less than ideal information and full analysis of individual resources and conditions.

Farmer Behavior and the Potential for IPM

If the above premise is correct, then part (and perhaps much) of the challenge to move IPM adoption beyond the current plateau is prompting producers to understand the value of quality data and apply it in more systematic and rigorous ways than they currently are doing when choosing pest-control strategies. The simple answer, but not necessarily a simple task, is to increase producers' awareness and change their attitudes. The Iowa studies suggest that at least on the surface attitudes are already in place. For example, when asked if savings and benefits from detailed record keeping justify the added time, cost, and effort incurred, there is strong agreement and very little disagreement (Petrzelka, Padgitt, and Wintersteen 1995). This finding is not altogether surprising nor different from most adults who accept the notion of healthier diets and regular exercise to better wellness but continue with behaviors that are quite counter to that end. Habit, the path of least resistance, enjoying existing behaviors, avoiding less-desirable activities, and the ability to rationalize and justify a given behavior are strong impulses. This occurs among farmers as well as the general public. Theories of cognitive dissonance, which postulate a tendency to resolve such discrepancies if they are pointed out, have some support in laboratory experiments, but certainly they leave a lot of variance unexplained.

For many producers, much of the rationalization limiting adoption of high management appears rooted in an economy of scale framework and the belief that time can be more profitably invested in expanded acreage production rather than refinements in current production practices. Such an outlook is consistent with personal work preferences and is deeply ingrained in a fairly pervasive "agri-culture" that values "bigger is better," "big iron," and "macho" approaches to production, including pest control. Moderating such values to give greater priority to information as a commodity is a slow process, and one that needs to be approached actively and persistently.

Learning how to do this needs to be taken seriously. Much can be learned by listening to farmers, and much can be learned from those whose livelihoods are dependent upon "closing the deal."

First, some observations from listening to farmers. In a series of open-ended conversations with farmers across Iowa about high-management systems for soil conservation and water quality protection (Imerman et al. 1996), six criteria for making decisions reoccur in the transcripts. With some caution, these topics can be inferred to be relevant to other high-management and environmental-protection systems. In rank order, the criteria were: profitability, yield stability, production compatibility, input cost containment, risk reduction, and environmental quality. A case can be made that IPM systems have advantages for each of these farmer-defined priorities. However, the advantages are not always apparent and must be reinforced on a regular basis. One factor apparent to the reader of the transcripts of these conversations is the well-established psychological principle that individuals interpret events in the context of their own worlds of experience and modify interpretations significantly from "objective" information that is presented (Schkade 1994). Also, casual observation or an anecdote often takes on equal status (reliability, accuracy, and generalizability) of more rigorous scientific data unless the fallacy is confronted, something that is unlikely to happen with mass-media and passive-education strategies. Staff of the Iowa Model Farms Project have been perplexed by the discrepancy between profitability (as shown from project records) and perception of some of the project coordinators. This is both the frustrating and challenging aspect of being in the business of promotion and advocacy. Clearly, opportunities must be seized. As with other kinds of changed behavior, when appropriate, reinforcement should be offered, and discrepancies should be made obvious. Most public servants have been reluctant to be so bold as to do the latter, however.

This observation leads to a related final point, the necessity to overcome fear, reticence, and anxiety in asking for a commitment to action. Often, agency employees fear asking for a commitment from clients. This feeling is not altogether different from an adolescent asking for that first date or dance. Agency staff rationalize that "our role is to educate," "to provide technical assistance," or "to point out alternatives," and not to promote or recommend. To not bring to closure a decision and instead follow the path of least resistance is not worthy of a "change agent." And, it is not consistent with the expectations increasingly being placed on staff roles in public agencies. The need to call for action and ask for a commitment became very real for NRCS in implementing conservation compliance. And it is very real if IPM is to be on 75 percent of managed acres in just five planting seasons.

Assessing Barriers to IPM Adoption

There is a wide and diverse research literature on the adoption of IPM. The purpose of this section of the paper is not to review or synthesize this literature, rather it is to draw out the practical lessons to be learned from this body of research. The intent is to provide practical guidelines to plant pathologists, entomologists, agronomists, and the many other professionals for whom IPM is an integral part of their career objectives.

The dominant perspective used in this paper is that of the grower, producer, or farmer. It is based on the assumption that one does not increase the use of IPM practices among this group unless one first understands how and why new practices are adopted, rejected, or modified.

First and foremost, it is critical to understand that adoption is a process. It is not a discrete, dichotomous event where one moves from nonadopter to adopter status as the result of a single decision. While colloquial language may characterize the adoption process as a binary event, in actuality it can encompass a series of identifiable stages or steps.

The initial stage is where the grower needs to become aware of a specific IPM practice or set of practices. This awarenes occurs in one of two ways. The individual may have a problem (e.g., pest losses or a feeling that excess funds are being spent on agrichemicals) and is seeking a solution, or some external party calls attention to a hitherto unrecognized problem (e.g., health or environmental problems derived from a reliance on agrichemicals) for which this party also holds a solution (IPM). While the distinction between these two situations is important for designing intervention or marketing programs, for now the important fact is that the grower becomes aware of something called IPM.

The grower will then seek knowledge about this practice to evaluate both the production and economic dimensions. This knowledge will take a variety of forms; from formal scientific research results to hearsay at local producer gathering places. Obtaining sufficient knowledge about the practice may be easy and straightforward, or it may be complex and difficult. The knowledge about the practice, positive, negative, and ambiguous, is constantly updated in an effort to transform it into information that is locally salient and decision focused. A producer may decide on the basis of this information that the practice will not work, is not cost-effective, or may be worth a try.

If the practice (or practices) being evaluated is conducive to division, then a producer may decide to try using it on a small-scale basis first. This trial stage allows growers to assess whether they can manage the practice, if needed forms of inputs and assistance are readily available, and if the practice will be profitable across a production cycle. Because of the dynamic interaction of pest cycles, weather, and actions of neighboring producers, the trial process may be extended through several production cycles. If the outcome of this small-scale trial is positive, a producer may decide to move to full-scale adoption. That is, apply the practice to all applicable acres. Of course, the converse is also true. The producer may decide at any time in this decision process to reject the practice and maintain traditional practices while looking for other feasible solutions.

Factors Influencing This Process

Adoption or rejection does not occur as an individual act isolated from the context in which it occurs. Instead, a number of factors influence both the outcome and speed of this decision process. These include the nature of the IPM practice, characteristics of the operation, infrastructure support, and managerial capabilities. There are a number of research generalizations that tell us that the complexity, divisibility, cost, and compatibility of the practice influence the speed and outcome of the adoption process. Characteristics of the operation also influence the adoption process. For example, larger, specialized operations are more likely to adopt at a faster rate than smaller, diversified operations. A critical element is the amount and quality of what can be called infrastructure support, including factors like the amount and nature of research being conducted in the public sector, the viability of private-sector information markets, cost-effective access to supporting materials and supplies, availability of quality labor or managerial expertise, and the lack of active opposition from local agrichemical suppliers.

Measuring Adoption

Measuring the adoption of IPM practices can be more complex than it sounds. At first glance, it appears to be nothing more than a question of whether a grower is or is not using a specific practice. Yet this simplistic view quickly changes as one begins to assess how it is being used, where it is being used, and the appropriateness of that use relative to actual pest conditions. Complexity aside, measuring adoption of IPM practices is the foundation of any viable IPM program. These IPM programs, in either the public or private sector, often have goals or objectives associated with them. Being able to measure adoption informs the public or shareholders to the extent the program is achieving these goals or objectives. Measuring adoption can also provide information on the efficiency of the IPM program. Just how many resources are being used to achieve certain levels of adoption is a question that any organization or firm needs to address sooner or later. For public sector organizations who must also address equity issues, the question of who is adopting these practices is important. For example, has the program focused on those with the greatest economic need, those with the greatest human-health risks, or those where there is the greatest potential for environmental damage? All these questions are important, and all are based on the idea of measuring adoption of IPM practices in a valid and reliable fashion.

The foundation of any science is describing, explaining, predicting, and possibly controlling variation. For behavioral or social scientists, the focus is on explaining variation in human behavior. Producers, growers, and farmers, contrary to common perception, are not a homogeneous mass. There is as much richness and diversity in farmer behavior as there is in the pests and pathogens associated with IPM practices. Because of this diversity, it is difficult to discuss adoption as if it were a singular concept. The bottom line is that the methodological sophistication found in the sciences underlying IPM programs needs to be matched by efforts to measure the adoption process. Discussing the methodological sophistication needed to measure IPM adoption, however, is not the focus of this paper. Nonetheless, it would be an omission not to at least mention these issues while describing practical considerations in addressing barriers to IPM adoption. It would be difficult to know if a barrier exists or has been overcome unless one also measures the adoption process.

Measuring adoption of IPM practices can occur at four different levels of measurement, each of which has its own advantages and disadvantages. These measures are not mutually exclusive, but are sequential and cumulative. That is, one has to move through the lower levels of measurement to obtain higher levels of measurement.

Measuring Adoption with Accounting Measures implies the use of many of the traditional methods used to count audience response to programming efforts. Counting the number of individuals who participate in a program, who receive a newsletter or other educational material, or who show up at field days or demonstrations are all examples of the accounting method of measuring adoption. This is the simplest measure of adoption but is also the lowest in terms of validity and reliability. The only weaker measure known is the “wild guess” relative to adoption rates.

Measuring Adoption with Proportional Measures is perhaps the most common method used in formal studies. Individuals are asked if they are using certain practices or engaging in specified behaviors. These dichotomous responses (e.g., yes or no) are then statistically manipulated in one of three ways: (1) Individuals are classified as adopters or nonadopters of IPM based on the proportion of yes to no answers; (2) Individuals are classified as to the level of IPM use according to some ordinal scale of measurement (e.g., low, medium, or high), again based on the proportion of practices used that are judged to be critical to IPM; or (3) the extent of IPM adoption is calculated by determining the proportion of applicable acres on which the salient behaviors are applied (e.g., individual is using IPM on 68 percent of all corn acres).

Measuring Adoption with Accuracy-in-Use Measures attempts to account for the appropriateness of the salient behaviors. This level involves some measure of the ecological setting of the adoption behavior as well as the timing of the behavior. For example, spot spraying a postemergent herbicide at reduced rates may be an appropriate IPM behavior depending on weed composition and pressure. This level of measurement involves measuring features about the pest population within site-specific settings and then comparing actual pest control behaviors relative to recommended behaviors before making a judgment on IPM adoption. The phrase "accuracy-in-use" can be used to describe this method. It implies that adoption is more than simply engaging in a certain behavior, that the precision or accuracy of that behavior relative to pest conditions should dictate how IPM is being used. This method differs from the proportional measure of adoption in that it also accounts for the nature and level of pest pressure or for the risk of significant crop damage if inappropriate actions are taken. This latter factor is especially important in high-value horticultural crops.

Measuring Adoption with Distributional Measures is the most complex in that it incorporates both the spatial and temporal dimensions of the behaviors. It is an ecologically based measure of adoption in that determining which behaviors can be classified as IPM is dependent on pest dynamics across space and time (not limited to a field/grove or a particular period during the production process). Spatial patterns of pest dynamics (e.g., life cycles and mobility patterns) are examined to determine appropriate behaviors at particular points in time. IPM is based on landscape assessments of habitat conducive to pests, the distribution of agricultural practices, and efforts to model pest dynamics within this setting. Intervention strategies are designed on the basis of this system or holistic analysis. While no studies could be found that used this level of adoption measurement, the advent of spatial position and digitizing technologies should facilitate the development of this method.

The advantages and disadvantages of these four levels of measurement are summarized in table 1. Other comparative dimensions could have been selected, but the objective was to provide a broad overview of each of these methods within the context of an IPM program. The intent is to illustrate the resulting differences as one moves across the levels of IPM measurement.

Barriers to IPM Adoption

There is a need to abandon the stereotype that adoption of IPM occurs among "progressive" producers while nonadopters are "laggards" or "traditional" farmers. Basing the rationale for the nonadoption decision on psychological characterizations of the target audience is inaccurate, nonproductive, and not supported by the research literature. The dominant theme of the following material is that producers often have very good reasons for why they are either unwilling or unable to adopt IPM recommendations. Rather than "blaming" these individuals for their nonadoption decision, more effort needs to be spent on assessing why this outcome occurs. Those promoting IPM practices need to recognize that growers frequently have very good and rational reasons for rejecting IPM recommendations. These reasons, in light of the title of this paper, can be called barriers to IPM adoption. Understanding the distribution and strength of these barriers among target audiences is the basis for accelerating the adoption of IPM practices.

Farmers do not adopt IPM practices for two basic reasons; they are unable or unwilling. These reasons are not mutually exclusive. Farmers can be able yet unwilling, willing but unable, and of course both unwilling and unable. These may sound like minor semantic distinctions, but the difference between a farmer being unwilling or unable is crucial when designing the appropriate remedial strategy. Accelerating the adoption of an IPM practice must be based on understanding why farmers are rejecting these technologies and recommendations. Are they unable, unwilling, or both?

Barriers: Being Unable to Adopt an IPM technique implies presence of an obstacle or situation where the decision not to adopt is rational and correct. The farmer is making a sound decision in rejecting an IPM practice because of this obstacle. The important point is that the farmer may be willing to adopt the practices, but for one

Table 1. Comparative Analysis of Different Measures of IPM Adoption

	Accounting	Proportional	Accuracy-in-Use	Distributional
Measurement issue	Any indicator of program participation	Dichotomous measure of IPM use or extent of use across applicable areas	When and how specific practices are used while accounting for appropriateness of action	Where specific practices are being used as defined by geographical or biological parameters
Unit of measurement	Individual	Number of practices used or percent of crop acres	Difference between actual use and recommended use	Spatial pattern (polygon) of use in a landscape
Cost	Low	Moderate	Moderate to high	Moderate to high
Ease of use	Easy	Moderate	Complex	Complex
Utility	Low for program justification and evaluation of effectiveness	Adequate to estimate level or extent of adoption of specific practices	Good for targeting to increase efficiency of an IPM program	Good for targeting to increase effectiveness of IPM programs
Validity	Low	Moderate	High	High
Sample frame	None: count program participants	Usually random sample	Random; population of targeted area; stratified or proportionate by IPM user	Spatial sampling based on geographical or ecological features
Required disciplinary mix	None, any discipline can manage	Typical leadership by one discipline with cooperation of other sciences	Multidisciplinary with complementary responsibilities among social and biological sciences	Interdisciplinary with issues and methods being developed concur-rently

or more of the following nine reasons is unable to make this decision. Each reason for inability to adopt is followed by a brief summary of the appropriate remedial strategy.

Information Lacking or Scarce. A farmer may be unable to adopt a practice because some of the basic information needed for a sound economic and agronomic analysis is missing. *Remedial Strategy: develop and distribute the necessary information to those needing it.*

High Cost of Obtaining Information. Even in our highly touted information age, the time, expense, and difficulty of obtaining site-specific information may be too high. Contrary to common belief, obtaining relevant information is not free to the farmer. *Remedial Strategy: increase accessibility and ease of obtaining the basic information for those needing it.*

Production System Too Complex with IPM. A defining characteristic of any production technique is its simplicity or ease of use. There is an extensive research literature that shows the complexity of a technology is inversely related to the rate and degree of adoption. *Remedial Strategy: redesign and simplify the IPM recom-mendations or encourage incremental adoption.*

IPM Practice Too Expensive. Investment, costs, and influence on net returns are major concerns of today's commercial farmer. Systems must be agro-nomically sound and have an affordable price tag. *Remedial Strategy: subsidize the adoption decision or redesign a less expensive system*.

Excessive Quantity or Quality of Labor Requirements. Land, labor, and capital still determine the nature of the farm firm. The labor requirements associated with an IPM technique must be perceived as commensurate with the capabilities of the farm firm. *Remedial Strategy: redesign the IPM technique to reduce labor requirements or subsidize the hiring of adequate labor*.

Too Short a Planning Horizon to Begin the Adoption Process. An IPM practice may be rejected by a farm firm because of the current planning horizon relative to the time associated with recouping initial investments, learning costs, or depreciation of capital investments. Many of today's farmers will not be farming in two or three years because of retirement and other transitional forces. Their making a long-term investment within the context of a short planning horizon is not logical. *Remedial Strategy: redesign the system for incremental adoption or subsidize a short-term unprofitable decision.*

Limited Availability and Accessibility of Supporting Resources. Few farmers adopt a new production or IPM practice without significant support. This support can take the form of local crop consultants or agrichemical dealers willing to take the risk of supporting practices not currently being used in their trade area, other farmers using these practices who are willing to share both successes and failures, and a USDA research and assistance network capable of answering farmer questions. *Remedial Strategy: build the capacity of local assistance networks to meet local demands. Target the development of local assistance networks in the areas needing them the most. Develop methods to promote IPM practices on the basis of need, not the ability to pay or past cooperator status.*

Inadequate Managerial Skills. As in the case of the physical resource base they manage, there is tremendous diversity among farmers. One dimension of this diversity is managerial skill. Too often IPM practices are designed for the average or above-average manager. Local assistance networks are also oriented to this group of farmers because of the performance and evaluation systems used in USDA. All this can create a situation where farmers with less-than-average management capabilities receive little or no assistance to build these skills. *Remedial Strategy: focus assistance and skill-building opportunities on those farmers needing them the most, not just the most receptive.*

Little or No Control over the Adoption Decision. It is common to view the farmer as some independent decision maker who "calls all the shots." The farmer, therefore, becomes the focal point of most efforts to transfer new practices. In many situations, however, a decision cannot be made without the approval of a partner, source of financial credit, landlord, or some other third party. These other interests must be convinced of the merits of an IPM technique. *Remedial Strategy: Determine who can make or has significant influence on adoption decisions and focus efforts on those persons or organizations. Also, recognize that an adoption decision is often a family decision, and therefore persuasion efforts need to address relevant family members.*

Barriers: Being Unwilling to Adopt an IPM practice implies that the farmer has not been persuaded that the practice will work or is appropriate for the farm operation. There are a number of reasons why this persuasion does not occur. Again, as in the case of the inability to adopt, many of these situations are beyond the farmer's control. Therefore, the farmer is making a correct decision in rejecting the practice. Until the correct form of persuasion is offered to the farmer, this land manager will remain unwilling to adopt. Six reasons for being unwilling to adopt with a synopsis of appropriate remedial strategies follow.

Information Conflicts or Inconsistency. A farmer may be unwilling to adopt an IPM practice because of inconsistency or even outright conflicts in the information about the practice. A farmer may hear that a IPM practice will increase labor requirements, increase risk, or narrow windows of opportunity to accomplish certain tasks. The farmer may also hear about the experiences of another local farmer who claims it requires less labor, does not influence risk, and has no influence on timing of activities. These types of divergent messages must be resolved in the farmer's mind. *Remedial Strategy: work to develop a consistent information base. Where legitimate differences exist, offer explanations of these differences.*

Poor Applicability and Relevance of Information. To make a sound decision, farmers need information that is applicable and relevant to their farms. Data from a neighboring state or even across the county may be judged as not meeting local conditions. To be convincing, these data must be adapted and made available relative to local situations. *Remedial Strategy: develop and distribute relevant information on a local basis.*

Inconsistencies Between Current Production Practices and the IPM Procedures. IPM practices do not always easily fit into existing production systems. In these cases, the general expectation has been that the farmer will adapt operations to meet the adoption requirements of the IPM practice. This case can be contrasted with a situation where a flexible technology is designed so that it can be adapted to fit into a farmer's operation. *Remedial Strategy: develop flexible IPM practices capable of being altered to meet unique farm conditions.*

Ignorance on the Part of the Farmer or Promoter of the IPM Practices. Ignorance is not a pejorative term. Instead, it implies a situation where an individual has not had the opportunity to learn. This ignorance could be surrounding the basic economic and agronomic facts of the IPM practice, or for change agents it could be a lack of sensitivity to the basic needs of a potential adopter. *Remedial Strategy: determine the actual (and not the assumed) assistance needs of the target audience; then design education and assistance programs based on farmers' needs, not agency or business expertise.*

Increased Risk (Real or Perceived) of Negative Outcomes. An IPM practice can increase the probability of a negative outcome in many ways. The complexity of a practice or system into which it is incorporated, importance of the timeliness of operations, and the interdependence of inputs can all increase perceived or real uncertainty and risk. Some farmers are simply unwilling to make a major decision under conditions of uncertainty, or where there is significant risk. *Remedial Strategy: redesign the IPM practice or address risk in two basic ways; either increase information so probabilistic outcomes can be calculated, or subsidize the farmer to take a risk.*

Belief in Traditional Practices. Although we often scorn traditional beliefs and practices in agriculture, let us not forget that those "traditional" farmers continue to survive in today's competitive environment while thousands of their "innovative" or "progressive" neighbors have gone out of business. Some farmers are unwilling to change because those traditional practices represent the least risk in dynamic agricultural markets. *Remedial Strategy: demonstrate not only that the new way (use of IPM practices) is better than the old way but also that the new way does not increase risk for the farm operation.*

Putting It All Together: Assessment and Targeting to Accelerate the Adoption of IPM

One can make at least three general observations from the foregoing lists of why farmers are either unable or unwilling to adopt IPM practices. First, increasing the adoption of IPM practices is dependent on first addressing reasons why farmers are unable to adopt. Once these impediments are removed, then it is a question of persuading the farmer from being unwilling to adopt.

Second, many of the factors causing farmers to be unable or unwilling to adopt are beyond their control. Blaming the farmer for not adopting IPM practices is not only erroneous in many cases, it is also hypocritical. Instead of always focusing on the farmer, more attention needs to be given to our efforts in understanding and addressing the many reasons why farmers are unwilling or unable to adopt. In many cases it is not so much a "farmer failure" as it is a "system failure."

Third, broad-scale use of any one or even several of the remedial strategies suggested is doomed to failure. A “shotgun” approach to using technical, financial, or educational assistance is not the answer. Instead, considerably more effort needs to be spent trying to understand the reasons why a farmer may be unable or unwilling to adopt. Based on spatial distributions of those reasons, one should be able to target specific types of assistance in a format compatible with the capabilities of the target groups. The promotional strategies that worked for the early adopters will not be as effective with later adopters. If we want accelerated rates of adoption for IPM practices, then we must be as willing to accept new ideas and methods as we expect potential adopters to be.

One final observation is relevant to this topic. During the past 50 years, we have seen tremendous shifts in the structure of our agricultural system, significant gains in the science of detecting and explaining natural-resource problems, and extensive advances in both resource-management policy and the IPM practices supported by these programs. But despite all these advances, we are still in the “horse and buggy” days of understanding and meeting farmers’ needs as defined by the farmer. Instead of using the sophisticated communication campaigns and marketing strategies commonplace in agriculture's private sector, we continue to rely on crude “educate, regulate, or bribe” tactics. Unless we begin to spend a little more time and effort trying to understand all the complex reasons why farmers are unable or unwilling to adopt, our aspirations for wide-scale adoption of IPM practices are destined to fail.

Social Influences on and Impacts of IPM

Building public support for integrated pest management (IPM) is essential. On the one hand, farmers need information and motivation to adopt IPM practices. On the other hand, public officials and citizens need to better understand and support farmers’ efforts to produce food with reduced chemical inputs. It is also important to anticipate and manage the social impacts of new farming practices, such as those associated with IPM.

In this section of the paper, we provide IPM professionals and others with three kinds of information and strategies that will make it easier to work effectively with a wide range of groups and individuals. First, we discuss how to build productive IPM partnerships. Second, we present proven techniques for managing conflicts, building consensus, and improving communication. Finally, guidelines are provided for assessing and managing the social impacts of IPM.

IPM professionals work within a larger community that includes colleagues from other disciplines, as well as a range of stakeholder groups. You need to understand the people, politics, and institutions in your community. Formal organizations, such as government agencies, bring individuals together to pursue goals they cannot achieve alone. Less formal groups also permeate a community. These include political leaders, community organizations, the media, and other stakeholders. Stakeholders include any individuals or groups who have an interest in or will in some way be affected by your IPM efforts. Farmers, environmental groups, government agencies, farm businesses, and recreational users are examples of stakeholders. Social customs and cultural values also influence IPM acceptance.

Several broader societal trends may influence IPM efforts. One important social trend involves shifting demographics, including urbanization (Hoban 1994). Most people today have little understanding of or appreciation for agricultural issues and problems. Political power and influence continue to shift away from the agricultural community toward nonfarm interests. The farm sector is expected to produce a cheap and abundant supply of food while reducing the use of chem-icals, water, and land. On a related point, as people move from urban to rural areas, conflicts can arise over issues like pesticide use, livestock waste, and other perceived risks. How the agricultural sector responds to these and other social issues will influence future policies and programs.

Another important trend is the development of broad-based and strong public support for environmental quality (Dunlap and Catton 1979). A profound societal shift has occurred in people's views about the environment (Buttel 1987). Most people now hold an environmental world view and have values that will support IPM. The public has grown more concerned about environmental and food-safety risks (Hoban 1991). People are demanding a greater voice in decisions about risk management. The public wants a risk-free world. Most people rely on intuitive risk judgments (typically called risk perceptions) rather than on scientific data. Their information comes largely from the media. People are also very concerned about indirect risks, such as impacts on quality of life, property values, and future generations. Because many influential political leaders have the same perceptions as other citizens, political decisions are often made on subjective grounds, as well.

Building IPM Partnerships

The human or "people" aspects of IPM have an important influence on the success of your efforts. Successful IPM requires partnerships among a number of different individuals, groups, and organizations. Through partnerships, people and organizations work together cooperatively toward a common goal. Partnerships allow for local development and ownership of solutions, which can heighten community support for IPM.

Farmers and landowners are vitally important because that is where the action takes place. Local businesses (including input dealers, banks, and consultants) influence adoption of IPM. Various government agencies provide information, as well as technical and financial assistance. They also have expertise in farm planning and management. Local elected officials are also vitally important because they provide political support. Other partners, including the media and teachers, can help with education and information efforts.

Partnerships are the backbone of effective natural-resource management (Hoban 1992). Partnerships can result in more efficient use of staff and financial resources. Partnerships foster a spirit of collaboration and cooperation. They can promote fairness and minimize the potential for negative social and economic impacts. Most importantly, partnerships lead to more creative and acceptable ways to protect natural resources. This is particularly true when a broad range of disciplines are involved with IPM efforts.

Partnerships do have some disadvantages. It takes time and skill to create successful partnerships. Maintaining motivation and enthusiasm is another challenge, especially if results do not happen quickly. You need to identify all the relevant stakeholders, then persuade these partners that their efforts are needed. As you build local partnerships, you will encounter these and other challenges. Keep in mind, however, that the benefits of partnerships will usually far outweigh the disadvantages.

Approaching Partnerships Positively. Success depends on involving the right mix of people and organizations in your partnership (Buckholz and Roth 1987). You will need to find people to play a number of roles. Some partners will need to have technical expertise. Some will need coordination and communication skills. It also will help if some partners have political connections or public-policy expertise. As you look around your community, you will find a number of different private and public groups who have a stake in the farming community and/or the environment. Each situation is unique. It is possible to outline several approaches that have been identified for building team performance (Katzenbach and Smith 1993).

- Select partners based on skills, not personalities. Your partnership will need technical, problem-solving, and interpersonal skills. Find the right people, and the partnership will be a success. It will also be important that partners have a spirit of cooperation.

- Establish a sense of urgency and direction. All partners need to believe in a worthwhile purpose. They also want to know what is expected of them. This will build commitment to the partnership and promote success.
- Set ground rules. You will need to set expectations related to meeting attendance, constructive feedback, and other expected contributions. Such rules encourage commitment, cooperation, and trust.

- Start with short-term tasks that have a good chance for success. First impressions mean a lot.

Be sure early projects are realistic and will be "winners." This will build confidence and positive momentum for your partnership.

- Challenge the group regularly with fresh information. New information that you will be gathering as a partnership will help to better understand your situation and improve your effectiveness. New facts often motivate people to action.

- Spend enough time together. It will take time to get your partnership working effectively. Spend time (outside of meetings if possible) to get to know each other and become more comfortable working as a partnership.

Building Consensus Among Partners. Partnerships work best with consensus decision making. The consensus approach offers a number of advantages (Carpenter 1990). First, it helps individuals learn about each other and gain new insights about important issues. Second, consensus decisions are generally better because they reflect the concerns of all parties involved. Third, when people have worked together to understand issues and develop solutions, the outcome is much more acceptable. Fourth, consensus usually leads to faster implementation of decisions (once they are reached) because resistance will be lower. Finally, the consensus process has the longer term benefit of building trust among the partners. The consensus process is most appropriate when issues are complex and negotiable (Susskind and Cruikshank 1987). Effective consensus decisions share the following characteristics:

- Participation is inclusive. All major interests are identified and brought together.

- Participants educate each other. They spend time discussing the history of the issue, their perceptions and concerns, and ideas for solutions. They help plan activities and offer suggestions to make them more effective.

- A common definition of the problem is used. Participants discuss and agree on a constructive definition of the problem.

- Multiple options are identified. Participants seek a range of options to satisfy their respective concerns and avoid pushing single positions.

- Decisions are made by mutual agreement. Participants do not vote; but modify options until everyone agrees that the best decision has been reached.

- Participants are responsible for action. They identify methods for implementing solutions and then work together to promote and monitor implementation.

Obstacles to Partnerships. Despite the best intentions, partnerships are often difficult to establish and maintain (Scholtes and Associates 1988). It is important to recognize and overcome obstacles to partnerships (Hoban 1992a).

- They lack time or other resources. The people in the partnership will also have other commitments. They may view group activities as an unimportant use of their time. Related to this may be other real or perceived costs of partnerships.

- Levels of commitment or interest are low. This can happen if the effort gets bogged down or members are not given enough interesting tasks to do along the way. It also reflects the fact that some members give joint efforts low priority.

- Individualism and elitism is evident. In many respects the idea of working together is contrary to our cultural beliefs in self-sufficiency and competition. People tend to feel it is a sign of strength to be able to solve their own problems. Some people or organizations seem to have one way of doing things and are unable to adapt to change.

- Concern is expressed about loss of autonomy or recognition. People (especially those who represent organizations) worry that partnerships mean a loss of freedom or control over their own activities. Some also worry they may not get enough credit for the work they do within a partnership.

- Goals or missions conflict. Partnerships generally involve diversity in members, including private businesses, public agencies, and citizen groups. These different organizations can have different goals and expectations for the partnership. In fact, some see partnerships mainly as a way to pursue their own agenda.

- Some participants dominate or feuds break out. Some members (often those with authority or expertise) have too much influence over a partnership. Such "experts" can discourage discussion or criticize others' ideas. Partnerships can become battlefields for individuals who have their own feuds or past problems.

Leadership and Coordination

Effective partnerships do not just happen. They depend on coordinators or leaders that emerge from the group (Morrison 1994). As an IPM professional, you may need or want to serve in such a role. Coordinators play some of the same roles as a traditional leader. They do not, however, assume the same control or responsibility as a formal leader. Effective coordinators have a number of important responsibilities. They generally catalyze activities and keep the partnership moving. The coordinator handles, or asks someone to handle, administrative responsibilities (such as preparing reports). This includes calling and conducting meetings.

Effective coordinators have certain characteristics (Scholtes and Associates 1988). They are interested in the group's issues or concerns. Coordinators understand and are sensitive to the social and political situation. Good communication and group interaction skills are also important. Effective coordinators are respected as knowledgeable and fair. They are also able to share responsibility and credit with others in the partnership. Coordinators can help promote compromise and make trade-offs. Good coordinators should be patient, creative, and flexible.

Effective Coordination. Partnerships rely on a skilled coordinator to get the partnership started and to keep it moving. Coordinators should serve as catalysts for the group's decisions and actions. They should not, however, make decisions for the group. This neutral role implies six leadership qualities that are helpful for effective coordination (Katzenbach and Smith 1993):

- Keep the purpose, goals, and approach relevant and meaningful. Coordinators should use their own skills and perspectives to help members of the partnership determine, clarify, and commit to the group's goals. They can inspire appropriate actions, but should not try to move the partnership in any particular direction.

- Build commitment and confidence. The coordinator must understand and try to balance the needs and interests of both individuals and the overall partnership. Positive and constructive feedback helps make the partnership more successful.

- Strengthen the mix and level of skill. Effective coordinators recognize and build on the strengths and skills of individual members of the partnership. Effective partnerships depend on having an appropriate balance of technical, interpersonal, and other skills. The coordinator ensures that all the necessary skills are available for the partnership.

- Manage relationships with outsiders, including removing obstacles. To be effective, partnerships often interact with other groups in the local area. Coordinators often have the responsibility of ensuring that the important external relationships are developed and maintained. Such responsibility may be shared with other members of the partnership.

- Create opportunities for others. Coordinators should not try to do everything themselves. They must provide opportunities for individuals if the partnership is to grow and work effectively. This involves attention to empowerment and delegation.

Understanding Communication. Partnerships are built upon open and ongoing communication (Hoban 1992b). To truly communicate, people must come to a shared understanding. Communication is a two-way process; listening is just as important as speaking. Communication is a skill that can be improved. The following are some general strategies for improving communication with others in your partnership (Williams 1983):

- Look for common ground. Find shared values. Consider shared personal experiences. Be willing to accept differences in perceptions and opinions.

- Find out about others. Learn about others' interests and needs. Consider their perspectives. Let others express themselves freely.

- Attack problems, not people. Do not waste time on personal hostility. Make other people feel good. Avoid criticism and put-downs.

- Give and get respect. Show respect for others' opinions. Put yourself in the other person's shoes. Be responsive to emotions. Speak with confidence, but remain tactful.

- Be explicit and clear. Share your ideas and feelings. Pay attention to nonverbal communication. Select words that have meaning for your listener.

- Proceed slowly. Present one idea at a time. Check for understanding and acceptance of each idea before moving on to the next. Speak in an organized and logical sequence.

- Use the five "Cs" of communication: clarity, completeness, conciseness, concreteness, and correctness.

Understanding Conflict

Most of us experience conflict. Conflicts result from diversity within our society (Susskind and Cruikshank 1987). Individuals and groups differ in their attitudes, beliefs, values, and needs. Conflicts can arise because people perceive shortages of important natural or social resources. Conflicts also arise out of past rivalries and personality differences. Conflict is a natural process that is not always negative (Carpenter and Kennedy 1988). In fact, conflict can even be healthy if it is effectively managed. Conflict provides opportunities for growth and innovation. Also, conflicts may indicate that timing is not yet right for a decision or that additional information is needed.

Conflict management is successful when parties come to a resolution that meets both individual and group needs (Fisher et al. 1991). Successful conflict management and negotiation aim toward achieving consensus. The goal is for all parties to "win" by having at least some of their needs met. Most of us have experience with conflict management and negotiation in private disputes (for example with a salesman over the price of a product, among family members, or with our employer). Public conflicts that may arise from issues (such as environmental quality) are like private disputes, but are also different in several important respects (Carpenter and Kennedy 1988). They generally involve a complicated network of interests and a complex set of issues. Also, procedures for resolving public conflicts are not as standardized.

Ingredients of Conflicts. Conflicts often result because people are different. In dealing effectively with conflict, the best approach is to understand and build on the differences to come up with new ideas. Differences may lead to conflict in several areas (Weeks 1992):

- Needs: Needs are essential to our well-being. Conflicts arise when we ignore others' needs, our own needs, or group needs. Conflicts may also arise when our ability to meet needs is blocked by another person or outside situation.

- Perceptions: People interpret reality differently. They have different perceptions of the severity, causes, and consequences of problems. Conflicts arise from misperceptions or different perceptions.
- Power: How people define and use power has an important influence on the number and types of conflicts they have, as well as what methods they use to manage conflict. Serious conflicts arise when people use power to gain an unfair advantage.

- Values: Values are beliefs or principles we consider to be very important. Serious conflicts arise when people hold incompatible values.

Conflicts also arise when one party refuses to accept that the other party holds something as a value rather than a preference.

- Feelings and emotions: Many people let their feelings and emotions become a major influence over how they deal with conflict. Conflicts also arise because people ignore others' feelings and emotions.

Analyzing Conflicts. Before you attempt to manage conflict, it is important to analyze the nature and type of conflict you are dealing with (Carpenter and Kennedy 1988). The following sets of questions focus on the parties involved, the substance of the conflict, and possible ways to manage conflict:

- The parties involved: Who are the parties involved with the conflict? How are the parties organized, and what is their power base? Are the parties capable of working together? What are the historical relationships among the parties?

- The substance of the issue(s): How did the conflict arise? How are the main and secondary issues described? Are the issues negotiable? Have positions been taken, and if so, are there common interests? What information is available, and what other information is needed? What values or interests are challenged?

- Possible procedures for conflict management: Would consensus serve all parties? Are there external constraints or other influences that must be accommodated? What are the past experiences (if any) of the parties in working together? What is the time line for a decision? Will an outside negotiator be needed?

Conflict Management. Once you have a general understanding of the conflict, you can consider several alternatives for dealing with the conflict. There are five basic strategies for managing conflict (Dotson, et al. 1989). Each has its own appropriate uses and inherent problems.

- Competition involves high concern for one's own interests with less concern for the other parties. The outcome is "win/lose." This is a common approach that includes most attempts at bargaining. Competition is generally used when basic rights are at stake. Unfortunately, the conflict can often escalate, and losers may try to retaliate.

- Collaboration involves a high concern for one's own interests, matched with a high concern for the interests of the other parties. The outcome is "win/win." Collaboration is generally used when concerns for others are important. It is also generally the best strategy when the public interest is at stake. This approach also helps build commitment and reduce bad feelings. The drawbacks are that it takes time and energy. Also, parties may take advantage of the others' trust and openness.

- Compromise involves a high concern for one's own interests along with a moderate concern for the interests of other parties. The outcome is "win some/lose some." Compromise is generally used to achieve temporary solutions, to avoid destructive power struggles, or when time pressures exist. The drawbacks are that parties can lose sight of important values and long-term objectives. This approach can distract the parties from the merits of an issue and also create a cynical climate.

- Accommodation involves a low concern for one's own interests combined with a high concern for the interests of other parties. The outcome is "lose/win." Accommodation is generally used when the issue is more important to others than to you. It represents a "good will gesture." It is also appropriate when you recognize that you are wrong or outmatched by the other parties. The drawbacks are that your own ideas and concerns do not get attention. You may also lose credibility and future influence.

- Avoidance involves a low concern for one's own interests coupled with a low concern for the interests of other parties. The outcome is "lose/lose." Avoidance is generally used when the issue is trivial. It is also helpful when confrontation has the high potential for damage or more information is needed. The drawbacks are that important decisions may be made by default or not at all.

Social-Impact Assessment

Social-impact assessment (SIA) is an important tool for identifying and balancing different interests in a political climate. Freudenburg (1986) points out that SIA is a hybrid offspring of science and the political process. It emerged in response to society's increased concern over environmental degradation and the social consequences of change. Dietz (1986) defines SIA as the identification, analysis, and evaluation of social impacts resulting from a particular action. A social impact is a significant improvement or deterioration in people's well-being or a significant change in an aspect of community concern.

SIA can be particularly appropriate for dealing with conflicts where different groups hold competing values and incompatible interests related to the use of natural resources. Conflicts can arise in any situation where some groups or individuals benefit at the expense of other groups. Since those who benefit from a proposed action are often different from those who pay the associated costs, problems of equity arise (Wolf 1983). As Hester and Cortner (1983) explain, there is nothing new about conflict in natural-resource management. What is new is that resource conflicts are moving more into the local arena, conflicts are more intense and frequent, and most resource managers have not dealt with such conflicts.

SIA can promote conflict resolution by illuminating how benefits and costs will be distributed among various groups. SIA can help ensure that benefits and costs are more fairly distributed. To understand where resource-related conflicts may occur information is needed about: the interests most likely to be involved, the strategies these interests may use to push forward their positions; and the impacts of such conflicts on public agencies and other stakeholders (Hester and Cortner 1983).

Social-Impact-Assessment Processes. Identification of impacts requires imagination, creative thinking, and an understanding of the people being impacted (Dietz 1986). During the analysis, probabilities are assigned to possible impacts, with the use of quantitative and qualitative data, as appropriate. Finally, evaluation integrates the information from the identification and analysis stages into an overall image of the impacts resulting from the proposed action. By nature, SIA should be future-oriented by anticipating consequences before they occur. Future-oriented research allows some chance to mitigate negative impacts and to reduce conflicts among groups. Explicit comparisons are made between conditions as they are likely to be with and without a proposed action (e.g., a new policy, program, technology, or project).

Bryan and Hendee (1983) explain that SIA estimates how proposed policies, programs, or practices will affect people's lives. The goal is to help managers make better decisions. They provide some general principles as a useful framework for SIA:

- Focus on major concerns and issues identified through public participation, talking with local leaders, expert opinion, and experience in similar situations. Collect information on variables that accurately represent the identified issues and concerns. Recognize that social impacts can be positive or negative depending on the context in which they are viewed.

- Note that social effects can be direct or indirect. Investigation of social consequences should include immediate impacts, as well as indirect effects that may be subtle, but important. The appropriate methods and approaches for SIA will vary with the kinds and level of impacts anticipated. Flexibility is needed in the variables used, populations sampled, and geographic areas covered. Methods used to project, compare, display, and disseminate results should reflect the anticipated impacts. The area analyzed may vary with the proposed action and the social effects being evaluated. Before collecting original data, use all existing databases from various governmental agencies, media accounts, research reports, and direct observation. Given limited time, money, and staff, the general idea is often to gather as little new data as necessary.

- The format for reporting SIA depends on what is found. An interdisciplinary team should interpret the significance of identified social impacts.

Decisions can then be made as to the type and level of public participation in decision making.

- Social impacts may be subtle. The cumulative effects of individual management policies and practices may be very large. Communities adjust to change and adapt to social impacts, thus providing a continuing change in baseline conditions.

Types of Social Impacts. The first task of SIA is to define the key variables of interest. It is important to have a rationale as to why each is included in the analysis. The kinds of impacts that should be considered in a given SIA depend on the policy, program, or practice being considered (Dietz 1986). Strategies for measuring the major concepts and collecting the data are also important considerations. This section will summarize those that are most relevant for SIA of integrated pest management.

Not all groups or individuals are equally affected by a particular action. Schnaiberg (1980) stresses the importance of focusing on distributional impacts. Differential impacts occur because different people are affected in different ways at different times. Some groups lose, others gain, and most others fall somewhere in between (i.e., gaining in some ways, but losing in others). In fact, many people may be relatively unaffected by a particular action or non-action. Impacts must be broken out by location, income, occupation, ethnicity, and other features of groups who are disproportionately affected. Researchers need to focus on how actions redistribute resources, wealth, and/or negative impacts among communities, groups, and individuals (Freudenburg 1986).

Impacts can also be grouped according to the social unit or area affected by the action. Many proposed actions have limited impacts on the nation as a whole, but tend to have significant impacts on local communities (Dietz 1986). Different groups of individuals within a limited area will also be affected differently. Impacts can also be distinguished based on how direct or immediate their consequences are for the affected groups. Direct impacts are easier to identify and measure than indirect impacts, which often result from the direct impacts. Likewise, impacts can be seen as relatively short-term or long-term in their effects.

Social impacts vary in terms of objective visibility to the affected populations (Dietz 1986). Subjective impacts are those that are perceived by and of concern to those who are affected. It does not matter whether an outside "objective" analyst finds these impacts of major concern. Objective impacts, on the other hand, are considered significant by the outside analyst, whether or not such impacts are of concern to those groups or individuals directly affected. An effective SIA must identify, analyze, and evaluate both objective and subjective impacts.

A variety of impacts need to be considered in relationship to any changes in policies, programs, or practices. Conditions or impacts that are considered in a given SIA vary with the nature of the proposed action(s). In most cases, the main dependent variable for SIA should be changes in the overall quality of life as experienced by the impacted groups. Social variables, however, have generally been given less attention than economic factors. Social variables are not always recognized as important by decisionmakers (Freudenburg 1986).

Social-Impact-Assessment Methodology. The goal of SIA is to predict and evaluate the full range of social impacts before they occur. According to Wolf (1983) the "bottom line" question is "Who benefits, and who loses if a proposed action were to be implemented?" SIA is, in fact, a multimethod approach that requires researchers to draw selectively from the full range of social-science methods and techniques. Each situation has unique features that require careful selection of appropriate SIA methodologies. Most forms of social-science research can and have been applied to SIA. The relevance of two commonly used techniques will be described: expert-opinion panels and opinion surveys.

To determine the scope and significance of impacts, it is often helpful to tap the knowledge and interest of those most qualified and willing to lend their insight. Structured group processes (such as focus groups) can be used to identify, analyze, and evaluate both subjective and objective impacts (Dietz 1986). Such panels tend to be relatively inexpensive, flexible, and productive. Panels should include technical experts, social scientists, and individuals familiar with the concerns of the various impacted groups. Panels can set priorities for focusing scarce resources (time and money) on the most important types of impacts.

Survey research is a common element of most SIAs. Surveys provide insights into the beliefs, attitudes, and values of various groups regarding a policy, program, or practice under consideration. Values and attitudes represent important data for understanding and evaluating social impacts. How people perceive impacts can be at least as important as the actual impacts. Finsterbusch (1983) explains that surveys provide not only self-reported facts about respondents but also their inner feelings, attitudes, and opinions that cannot be systematically determined in any other way. Decisionmakers need to understand what people like and dislike, as well as how they will respond to alternative actions. Surveys can help establish priorities and assess attitudes toward alternatives. Information can be obtained about community needs and concerns, as well.

Conclusion

It is critical to remember that SIA takes place within political, social, and economic contexts. Interest groups will try to influence the course of SIA efforts and shape the action under investigation. Timing will, therefore, be of critical importance in effective SIA. There are several considerations in ensuring that SIA is included at the right point in the planning process so it can actually influence decisions (Dietz 1986). SIA should be used to identify key impacts at the beginning of the process. Next, SIA should be used to formulate alternative plans. Informal procedures can be very useful in improving plans. Once a set of policies and plans emerges, SIA can help evaluate and judge the proposals.

Integrated pest management efforts will sometimes encounter existing conflicts or even create conflicts among various stakeholders. Such conflicts have a number of important characteristics. They often involve issues about the distribution of costs and benefits. The individuals or groups who benefit from IPM may not be the same as those who pay the costs of changing practices. Natural-resource conflicts are often portrayed in terms of environmental protection versus economic benefits. Keep in mind, however, that IPM can result in both economic and environmental benefits.

Through partnerships and communication, conflicts can be managed so that all sides have at least some of their interests met. IPM professionals need to work with social scientists to better understand the perceptions, needs, and practices of producers. Such interdisciplinary partnerships can also foster more creative, effective, and equitable approaches to IPM planning, implementation, and evaluation.

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Assessing IPM Impacts: Summaries of Selected Papers

Session 1: IPM Adoption: Obstacles, Incentives, and Measurement

Introduction

The Clinton Administration's goal of achieving adoption of IPM practices on 75 percent of crop acreage by the year 2000 has focused new attention on measuring and evaluating the extent and impact of IPM adoption in the United States. Lack of consensus on what constitutes a core set of IPM practices along with data-availability problems have been major obstacles in measuring IPM adoption. The significant crop and regional variation in recommended IPM practices has frequently not been captured in past adoption studies, which often used a standardized list of practices to measure adoption. In addition, the introduction of new production technologies and practices, especially biointensive ones, will require changes in recommended IPM systems that may limit the usefulness of measuring the adoption of specific practices. Moving beyond simple measures of IPM adoption and impact is the focus of this selected-paper session.

Papers Presented

Carlson, Gerald A., and Michelle C. Marra, *The Role of Transgenic Crops in Future IPM Programs: An Economic Perspective*, Department of Agricultural and Resource Economics, North Carolina State University, Raleigh, N.C.

The authors of this paper examined the potential impact the adoption of transgenic crops may have on the adoption of IPM practices and techniques. The recent introduction of two transgenic crops [herbicide-tolerant crop varieties (HTCV) and crop seeds containing the natural insect toxin, *Bacillus thuringiensis* (B.t.)] could have a major impact on current soybean, corn, and cotton production practices. Because these new biotechnologies have the potential to significantly alter existing pesticide use (both quantity and product), increase yields, increase crop tolerance of certain herbicides (they will likely increase pest tolerance to B.t.), and change the use of other farm inputs (tillage practices, rotations, and insect and weed monitoring) their adoption could require significant changes in IPM systems and recommendations. The authors identified critical factors influencing adoption and diffusion of these new biotechnologies and related them to IPM-implementation projects dealing with the affected crops or regions. In their view, failure to account for the potential changes in production will result in IPM research and extension projects that are outmoded.

Coli, William M., and Margaret Christie, *Status Report on a Regional Project to Identify Barriers to and Opportunities for Greater Adoption of IPM*, Department of Entomology, University of Massachusetts, Amherst, Mass.

Coli and Christie provided details of an approach they used to develop site-specific definitions of IPM systems. This approach was used in the northeastern region of the United States to measure adoption of a suite of IPM systems for several important crops (apples, potatoes, strawberries, sweet corn, and spring bedding plants). The purpose of this multistate effort was to develop a scientifically valid approach for establishing a baseline; accurately capturing different degrees of adoption of IPM; and measuring environmental, public-health, and economic impacts. The process consisted of four steps: (1) describe IPM systems that are currently ready for adoption; (2) determine the extent of current IPM adoption with statistically valid techniques; (3) track important environmental, economic, or public-health variables and compare to the established baseline to estimate changes produced by the adoption of IPM practices and techniques; and (4) on the basis of knowledge gained from the preceding steps, prioritize the most critical research, extension, and training needs limiting greater IPM adoption.

According to the authors, stakeholder involvement in the process of establishing IPM definitions, goals, and evaluation criteria was critical to this approach. The diversity of issues (production possibilities, availability of IPM and other alternative production practices, environmental and public-health concerns, weather, pest pressures, etc.,) varied considerably within and among states. Developing program goals and evaluation criteria that are credible with a range of stakeholders and are scientifically valid required multistate, multiorganizational, and multidisciplinary teams.

Gianessi, Leonard P., and James Earl Anderson, *The Influence of Integrated Pest Management Programs on Pesticide Use*, National Center for Food and Agricultural Policy, Washington, D.C.

Methodological deficiencies of past IPM evaluation efforts are reviewed by the authors, and the claim that IPM has resulted in pesticide use reduction is challenged. The authors argue that many factors influence pesticide use, including changes in pest density and type, weather, changes in crop acreage planted, regulatory actions, and development of pest resistance. In a series of case studies of pesticide use changes attributed to IPM, the authors found that in some cases pesticide use was reduced; however, in others pesticide use either increased or the change was not attributable to the adoption of IPM but to other factors, such as the increased use of lower-rate chemicals. They argue for a more comprehensive approach to documenting the impact of IPM adoption on pesticide use. Specifically, they call for detailed documentation of pesticide use (e.g., number of sprays, pounds of individual active ingredients used, and cost of each spray) and the establishment of a baseline for more scientifically valid before-and-after (adoption of IPM) comparisons.

Szmedra, Philip, *The Adoption of IPM in Cotton: Some Issues Concerning Measurement and Evaluation*, USDA, Economic Research Service, Production, Management, and Technology Branch, Washington, D.C.

Efforts to evaluate adoption and impacts of cotton IPM were reviewed by Philip Szmedra. Cotton is an interesting case study because of the "maturity" of the IPM program. IPM research and extension programs encouraging cotton producers to adopt IPM have been in existence for several decades, and many practices associated with IPM programs have been adopted by a majority of cotton producers. However, as IPM systems become more sophisticated and biointensive, sharper delineations of adoption along a continuum will be needed. Szmedra argued that the level of IPM adoption inferred from survey results for a specific crop can vary considerably according to the definition of IPM chosen. This variation is particularly true when trying to differentiate low, medium, and high adopters. To arrive at a more accurate assessment of IPM adoption, Szmedra recommended:

1. a multidisciplinary approach to defining biointensive IPM by crop accompanied by a weighting scheme to better define the IPM-adoption continuum; regional variation necessitates that definitions reflect site-specific differences in recommended IPM practices; and
2. the development of survey instruments that capture sufficient information to identify where respondents are on the IPM continuum and the resulting impact on pesticide use.

Session 2: Health and Environmental Impacts of IPM: Measurement and Valuation

Introduction

Measuring the physical or biological impacts of IPM adoption is a major step in the process of impact assessment. However, the multiple vectors of concern and the probable tradeoffs between economic, environmental, and public-health objectives foster the need for an integrating framework for evaluating these tradeoffs. For example, the substitution of one type of pesticide product for another may reduce pesticide expenditures, improve farm profitability, and reduce potential surface-water pollution but increase worker, wildlife, and beneficial-pest exposure to toxic materials. One problem often encountered in assessing multiple impacts is that the economic value of changes in the environment and/or public health resulting from the adoption of IPM are not priced in the marketplace (e.g., the value of clean water, reduced exposure to toxic materials, rural landscapes, and reduction in pesticide use). Several approaches to integrated assessment are discussed in this session.

Papers Presented

Antle, John, Susan Capalbo, Donald Cole, Charles Crissman, and Richard Wagenet, *Integrated-Simulation-Model Analysis of Economic-*

Environment-Health Tradeoffs, Department of Agricultural Economics and Economics, Montana State University, Bozeman, Mont.

The authors presented a general approach to assessing quantitatively the economic, environmental, and human-health tradeoffs associated with the use of agricultural technologies and how conditions may be improved through the adoption of more sustainable practices (such as IPM). This approach was designed to account for key measurement issues that arise in agricultural-impact assessment. These issues included: the temporal and spatial variability of agricultural impacts; the need to integrate disciplinary models and data at a small scale or level of aggregation, such as the field scale, at which impacts can be reliably modeled; and the need to assess impacts at a large scale or level of aggregation, such as the regional or population level, for purposes of risk assessment and policy analysis.

Antle et al. discussed an application of this approach in a case study of the tradeoffs associated with pesticide use in the potato-pasture production system in the Andean highlands of Ecuador. The interdisciplinary research team collected data on field-level production, pesticide use, watershed pesticide leaching, socioeconomic characteristics, and health status (which included a clinical examination to test for pesticide exposure). These data were then used in three integrated simulation models to assess the economic, environmental, and health impacts of various alternative pest-management scenarios. The Antle et al. analysis indicated that there are large tradeoffs between production and environmental and human health risks and that improved pest-management technologies to reduce pesticide use can help mitigate these tradeoffs.

Blair, Aaron, *The Agricultural Health Study: A Prospective Study of Cancer and Other Diseases among Men and Women in Agriculture*, Occupational Studies Section, National Cancer Institute, Bethesda, Md.

Blair presented a summary of the Agricultural Health Study currently being conducted by the National Cancer Institute (NCI) in collaboration with the National Institute of Environmental Health Sciences (NIEHS) and the U.S. Environmental Protection Agency. To evaluate the linkage between agricultural chemical exposure in the development of cancer, neurological, and other chronic disease outcomes, the Agricultural Health Study has established a large prospective cohort that can be followed for 10 years or more. This study is being conducted in the states of Iowa and North Carolina.

The objectives of the Agricultural Health Survey include: (1) identifying and quantifying cancer risks among men, women, whites, and minorities associated with direct exposure to pesticides and other agricultural agents; (2) evaluating noncancer health risks including neurotoxicity, reproductive effects, immunologic effects, nonmalignant respiratory disease, kidney disease, and growth and development among children; (3) evaluating disease risks among spouses and children of farmers that may arise from direct contact with pesticides and agricultural chemicals used in the home, lawns, and gardens and from indirect contact, such as spray drift, laundering work clothes, or contaminated food or water; and (4) assessing current and past occupational and nonoccupational agricultural exposures through periodic interviews and environmental and biologic monitoring.

During the first year of a 3-year enrollment period, 26,235 people were enrolled, 19,776 registered pesticide applicators and 6,459 spouses of registered farmer applicators. Study organizers estimate that the total cohort in 1997 will include approximately 75,000 adult study subjects. Based on first-year enrollment, the composition of the survey should break down to 49,000 farmer applicators (62 percent of the cohort), 20,000 spouses of farmer applicators (24 percent of the cohort) and 7,000 commercial pesticide applicators (14 percent of the cohort).

Mullen, Jeffrey, and George Norton, *Economic Value of Environmental Benefits of Integrated Pest Management*, Department of Agricultural and Applied Economics, Virginia Polytechnic Institute, Blacksburg, Va.

The authors presented the results of their case study estimating the economic value of the environmental benefits of apple and peanut IPM programs in Virginia. The first step in their approach was to identify the risks posed by individual active ingredients to eight environmental and public-health categories: (1) groundwater; (2) surface water; (3) acute human health; (4) chronic human health; (5) aquatic species; (6) birds; (7) mammals; and (8) arthropods. They then assigned each pesticide to a risk category (high, medium, low, and no risk) for each of the environmental and public-health categories. Second, they defined the degree of IPM adoption and assessed the effects of IPM adoption on pesticide use by degree of adoption. Third, they estimated "willingness to pay" to reduce pesticide risks. These estimates are derived with contingent valuation (CV), a widely used (though controversial) approach to value nonmarketed goods. The authors used opinion surveys to ask respondents to assess the value of hypothetical goods or actions and to estimate the amount they would be "willing to pay" for those changes in real or potential risk.

The value of the environmental benefits obtained from the CV analysis were used to calculate the economic value of environmental benefits resulting from the adoption of IPM practices. The results of this study show that, in Virginia, the peanut IPM program reduced pesticide use but nonsignificant reductions in pesticide use resulted from the apple IPM program. The authors concluded that the peanut IPM program produced substantial environmental benefits but the apple IPM program produced no significant environmental benefits.

Owens, Nicole, Scott Swinton, and Eileen van Ravenswaay, *A New Way to Measure Farmer Willingness to Pay for Safer Herbicides*, Department of Agricultural Economics, Michigan State University, East Lansing, Mich.

Understanding the factors influencing a farmer's decision to use safer pesticides is the subject of the paper presented by Owens, Swinton, and van Ravenswaay. In this study, the authors proposed a method to develop estimates of herbicide demand and farmer "willingness to pay" for safer corn herbicides. The authors also examined the importance of prior knowledge of the health and environmental effects of herbicides, sociodemographic characteristics, and sources of information about production alternatives in influencing farmers' willingness to pay for safer herbicides. The method used in this study capitalizes on the fact that a well-defined market exists for atrazine. In a survey of Michigan farmers, respondents were asked to value "new" herbicides, similar to atrazine, but safer in terms of groundwater leaching potential, human risk of cancer, or toxicity to fish. Fifteen price combinations were derived from three different base prices of atrazine and five different price differentials for the safer formulations of the herbicide.

Preliminary survey results suggested that Michigan farmer's willingness to purchase safer formulations of atrazine appeared to be significantly related to the price difference over ordinary atrazine. For example, when the price differential between atrazine and a nonleaching substitute was zero, more than half the respondents indicated they would purchase the new formulation. However, when the price differential was $3.00 per pound, the percentage willing to purchase the nonleaching product fell to roughly a third. The fact that most respondents were not familiar with many of the health and environmental effects of atrazine may explain some of the observed lack of interest in safer herbicide formulations. While 60 percent of respondents reported hearing about the potential for atrazine to leach, less than half knew it is a possible human carcinogen; can irritate the skin and eye; and is slightly toxic to fish, mammals, and birds. When presented with potential health and environmental effects, respondents often doubted their validity. Survey results indicated that respondents mainly relied on product labels and herbicide dealers for health and environmental information about atrazine.

Session 3: Pesticide Use, Productivity, and Alternatives

Introduction

The role of IPM in contributing to reduced pesticide use has been debated for two decades. Case studies presented at the Third National IPM Symposium/Workshop and in other fora have reported mixed impacts of pesticide use resulting from IPM adoption. This is not surprising given that the scouting methods, economic thresholds, and other IPM tools that have been developed and implemented over the past several decades, primarily for managing major insect pests, have been aimed at improving the efficiency of insecticide use but not necessarily reducing use. In this session, methodological issues involved in measuring changes in pesticide use and factors influencing pest-management choices are discussed. In addition, some empirical results of IPM adoption on pesticide use are reported.

Papers Presented

Ferguson, Walter, Jet Yee, and Mike Fitzner, *Nonchemical Pest- and Nutrient-Management Practices: Limitations to Adoption and Policy Options,* Economic Research Service, Washington, D.C.

During the past decade, the role and importance of crop consultants in influencing farmers' pest- and nutrient-management decisions has expanded. Farmers faced with complex and information-intensive pest-management decisions have turned in increasing numbers to paid consultants for their site- and time-specific recommendations. Ferguson, Yee, and Fitzner presented the results of a 1994 survey of independent crop consultants. The survey explored consultants' perceptions of the level of adoption by farmers of nonchemical pest- and nutrient-management practices and major factors aiding and limiting adoption. Independent crop consultants surveyed indicated that the major limitations to adoption of IPM practices are lack of viable nonchemical tactics, potential lower yields, higher production costs, higher management skills required, lack of information, and lower crop quality.

Hubbell, Bryan, and Gerald Carlson, *Insecticide Selection, Application Rates, and Application Frequencies: Is IPM More Than Total Use Reduction?* Department of Agriculture and Applied Economics, Georgia Station, Griffin, Ga.

Hubbell and Carlson examined the often-claimed proposition that IPM adoption results in pesticide use reduction. In their review of the literature the authors found limited empirical support for IPM's claim of pesticide-use reduction. Further, they discussed why total pounds or expenditures for pesticides may not be the appropriate measure given, the importance of toxicity, persistence, and application rates per acre in determining economic and environmental outcomes. The analysis examined the impacts of four IPM practices: scouting, beneficial insect management, pheromones, and pruning. Using data on U.S. apple growers, the authors estimate three insecticide "component" models and examine the impacts of IPM use on selection of low-rate, low-toxicity insecticides and per-acre application intensity of selected insecticides.

The authors found that the IPM practices studied had a significant impact on selection of insecticide active ingredients and that certain practices significantly affect application rates. However, the selection effect appeared to be toward more specific, highly effective products rather than toward low-rate, low-toxicity insecticides. Adoption of IPM practices did not significantly affect application frequencies, suggesting that IPM adoption may not lead to significant reduction in insecticide quantities used in apple production. The authors argued that if reduction in pesticide toxicity or quantities is the desired outcome, other mechanisms, such as input taxes, may be needed to encourage growers to use safer, low-rate insecticides.

Lichtenberg, Erik, and Rae Zimmerman, *Adoption of Alternative Pest-Management Practices and Pesticide Use* in *the Mid-Atlantic,* Department of Agricultural Economics, University of Maryland, College Park, Md.

The authors examined the complex set of factors influencing farmers' willingness to adopt nonchemical pest-management practices. The study is based on a recent survey of corn and soybean farmers in New York, Pennsylvania, and Maryland. The in-depth survey elicited information on individual farmers' pest-management practices, including several measures of pesticide use (i.e., type of pesticide, number of acres treated, number of applications), use of nonchemical means of control, characteristics of farm operation, farm-level economic indicators, demographic and human-capital indicators, health problems related to pesticides, and attitudes toward health and environmental problems from pesticides. The authors developed a model to assess the adoption of nonchemical controls as a discrete-choice problem where adoption is a function of characteristics of the farm operation, human capital and demographic factors, experiences with health problems from pesticides, and attitudes toward health problems and wildlife injury from pesticides.

Session 4: Interdisciplinary Modeling: Issues and Examples

Introduction

A critical component of efforts to increase the adoption of IPM is the availability of valid and timely information on the cost-effectiveness of IPM compared to conventional agricultural practices. For adoption to occur, producers must be convinced of the cost-effectiveness, profitability, and/or environmental benefits of proposed pest-management alternatives. In addition, from a larger perspective, society must be able to weigh the potential tradeoffs among production, environmental, and public-health objectives. Methodological and data limitations resulting in part from the complexity and diversity of U.S. agroecosystems have contributed to the difficulties encountered in previous attempts to measure the cost-effectiveness of IPM methods. In this session, different methodological issues involved in estimating pesticide productivity and cost-effectiveness are discussed, and alternative approaches proposed.

Papers Presented

Chambers, Robert, and Erik Lichtenberg, *Econometric Evaluation of IPM in Maryland Field Crops*, Department of Agricultural and Resource Economics, University of Maryland, College Park, Md.

Information about the cost-effectiveness of IPM methods is critical for increased adoption. However, shortcomings encountered in past attempts to estimate economic impacts have included how in-season production adjustments and substitutions were modeled, and the reliance on experimental plot conditions that frequently failed to reflect in-field conditions. Chambers and Lichtenberg outlined an econometric approach to estimating pesticide productivity. They illustrated the elements of their method with data from a detailed, farm-level survey on pest management practices, pest conditions, and crop yields of Maryland field-crop producers. They discussed how this approach could contribute to a better understanding of issues related to IPM promotion efforts, such as the impacts of IPM programs on pesticide productivity, the relative cost-effectiveness of IPM and conventional pest-management approaches, and whether IPM results in reduction in pesticide demand by profit-maximizing farmers.

Du, Fang, *Production Function Estimation with Pest-Tolerant Response*, Department of Agricultural Economics and Marketing, Rutgers University, Cook College, New Brunswick, N.J.

The author explored methodological issues involved in developing a production model of pesticide use with pest-tolerant response to examine pesticide efficacy and profitability. Incorporating pest-tolerant responses in modeling pesticide productivity represents an improvement over previous modeling efforts because it addresses the fact that plants will tolerate some quantity of injury from pests without reducing marketable yield. Thus, it is important to distinguish between an input's direct contribution to output (productive) and one that contributes indirectly to output (protective). This study improves upon previous attempts at specifying pesticide production functions by differentiating between productive and protective inputs. The author does this by incorporating into the production function an "abatement function" that includes pesticide level, initial pest population, and pest-tolerant response. Field data are used to test the production model.

Lamp, William, Erik Lichtenberg, David Liewehr, and Lester Vough, *Joint Use of Intercropping and Pesticides to Control Leafhopper on Alfalfa*, Department of Agricultural Economics, University of Maryland, College Park, Md.

William Lamp, Erik Lichtenberg, David Liewehr, and Lester Vough examined five different levels of insecticide application rates to alfalfa plots grown with and without oat intercropping. Oat intercropping is a promising nonchemical means of leafhopper control. Data from a set of experiments were used to evaluate the impact of oat-alfalfa intercropping on the profit-maximizing level of pesticide treatment of leafhopper and on the resulting quantity and quality of forage. These data were used to estimate the parameters of (1) a model linking quantity and quality of output with leafhopper densities in the presence and absence of the oat intercrop and (2) a model representing leafhopper densities as a function of the insecticide application rate. These models were combined with output and insecticide prices to calculate the profit-maximizing insecticide application rate and associated threshold leafhopper density to evaluate the cost-effectiveness of the oat intercrop relative to reliance on chemical means of control.

Swanton, Clarence, and Stephen Murphy, *Weed Science Beyond the Weeds: The Role of Integrated Weed Management (IWM) in Agroecosystem Health*, Department of Crop Science, University of Guelph, Guelph, Ontario, Canada.

Swanton and Murphy made the case for moving beyond descriptive approaches to integrated weed management (IWM) (i.e., the impact on yields and weed interference of different management strategies, such as tillage, cover crop, planting patterns, etc.,) to predictive approaches that estimate future weed problems and the economic risks and benefits of interventions. The authors argued for using predictive IWM approaches that focus on agroecosystem health and integrate biophysical, social, and economic concerns. Two benefits of linking IWM to agroecosystem health were identified by the authors: (1) predictive models within IWM can be incorporated into larger agroecosystem models and (2) the relevance and benefits of IWM should become clearer to the public and government.

Session 5: Economic Impacts of IPM Adoption: Case Studies

Introduction

Methodological and empirical issues encountered in estimating economic impacts of IPM adoption, both *ex ante* and *ex post*, are tackled in this session. The diversity of IPM systems, research questions, and data
availability engender a variety of methodological approaches to measuring impacts of IPM adoption.

Papers Presented

Fernandez-Cornejo, Jorge, *The Microeconomic Consequences of IPM Adoption with an Application to the Case of Tomato Growers*, Economic Research Service, USDA, Washington, D.C.

The author presented a method for calculating the impact of IPM on pesticide use, yields, and farm profits and then applied this method to the case of IPM adoption among fresh-market-tomato producers in eight states. Results of this study indicated that, among fresh-market-tomato growers, adopters of IPM for insects and IPM for diseases applied significantly less insecticides and fungicides respectively than did nonadopters. In this study, IPM adoption for insects and diseases did not have a significant effect on yields and only a small impact on profits. Other factors found important in determining pesticide demand were pesticide prices, farm location, contractual arrangements for the crop, and farm size.

Hamming, Michael, Annu Rauf, Gerald Carner, and Haiyue Nie, *Impact of Widespread Adoption of Integrated Pest Management by Shallot Growers in Indonesia*, Department of Agricultural Economics, Clemson University, Clemson, S.C.

Hamming, Rauf, Carner, and Nie estimated the economic impact in Indonesia of mechanical versus chemical spray applications to control *Spodoptera exigua*, a major insect pest of shallots. Field studies conducted on shallot production in West Java in 1993/1994 collected basic economic information on costs and returns from shallot production, detailed information on pest control methods, and attitudes of farmers regarding some of the key issues of IPM. The economic information was used to construct a statistical model of the shallot production function. Results of the econometric production function analysis showed that all inputs made statistically significant and positive contributions except chemical fertilizers, foliar fertilizer, and pesticides. Data indicated that hand picking alone provided control as effective as insecticide use and hand picking combined. The estimated economic impact of adopting mechanical pest control was calculated as savings from eliminating all insecticide use and reducing sprays to occasional fungicide applications, a potential annual saving countrywide of $46.9 million. If health impacts of pesticide use by shallot growers were included, savings in lost productivity would be even greater.

Jans, Sharon, and Jorge Fernandez-Cornejo, *A Case Study on the Impact of IPM for Oranges in Florida and California*, Economic Research Service, USDA, Washington, D.C.

The authors analyzed the impact of IPM adoption on pesticide use, yields, and producer profits for Florida and California orange growers. In this study, no significant differences were found to exist between IPM adopters and nonadopters when measuring yields, profits, and the number of insecticide applications. The analysis also indicated that nonadopters were more likely to be engaged in off-farm work compared to IPM adopters. The authors argued that the intensive management requirements of IPM for orange production may be an important barrier to IPM adoption.

Scorsone, Eric, *Economic Evaluation of a Proposed Price-Flexible Action Threshold for Tart Cherries*, Department of Agricultural Economics, Michigan State University, East Lansing, Mich.

In this paper, the author examined the performance of a new action-threshold strategy for tart cherries in Michigan. Prices of tart cherries fluctuate widely from season to season, and this uncertainty is not captured in the currently available price-static decision rule underlying action thresholds. Scorsone compares price-static and price-flexible action thresholds for tart cherries with a bioeconomic simulation model. Results of his assessment indicate that the proposed price-flexible action threshold could potentially improve economic performance over the price-static action thresholds and non-IPM strategies.

Swinton, Scott, Leah Cuyno, and Frank Lupi, *Factors Influencing the Adoption of IPM for Corn Rootworm in Michigan*, Department of Agricultural Economics, Michigan State University, 202 Agriculture Hall, East Lansing, MI.

The authors of this study examined factors influencing adoption of three alternative pest management practices (scouting, crop rotation, and reduced insecticide rates) to reduce corn rootworm insecticide use among Michigan corn producers. In addition to explanatory variables often found in adoption studies (farm management practices, personal characteristics of adopters, physical environment and institutional environment), the authors included variables capturing producers' perceptions of financial and environmental risk, yield loss expectations, and sources of pesticide information. The statistical analysis used by the authors included probit and tobit estimation procedures.

Results of the analysis indicated that general management practices, personal characteristics, physical environment, and institutional environment all play a role in determining adoption of the three alternative pest management practices. The analysis also showed that farmer expectations about yield loss in a normal year and source of pesticide information are key variables in explaining adoption of reduced insecticide practices. In contrast the financial and environmental risk variables were not significant in affecting adoption of reduced insecticide practices. The authors concluded that educational programs to inform farmers about the likelihood of economically damaging rootworm infestations might be warranted given the importance of yield loss expectations in influencing reduced insecticide use. In addition, farmer reliance on industry sources for pesticide information suggested that agribusiness should be included in these educational efforts.