Risk Assessment for National Natural Resource Conservation Programs

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Abstract

This paper reviews the risk assessments prepared by the U.S. Department of Agriculture (USDA) in support of regulations implementing the Conservation Reserve Program (CRP) and Environmental Quality Incentives Program (EQIP). These two natural resource conservation programs were authorized as part of the 1996 Farm Bill. The risk assessments were required under the Federal Crop Insurance Reform and Department of Agriculture Reorganization Act of 1994. The framework used for the assessments was appropriate, but the assessments could be improved in the areas of assessments endpoint selection, definition, and estimation. Many of the assessment endpoints were too diffuse or ill-defined to provide an adequate characterization of the program benefits. Two reasons for this lack of clarity were apparent: 1) the large, unprioritized set of natural resource conservation objectives for the two programs and 2) there is little agreement about what changes in environmental attributes caused by agriculture should be considered adverse and which may be considered negligible. There is also some “double counting” of program benefits. Although the CRP and EQIP are, in part, intended to assist agricultural producers with regulatory compliance, the resultant environmental benefits would occur absent the programs. The paper concludes with a set of recommendations for continuing efforts to conduct regulatory analyses of these major conservation programs. The central recommendation is that future risk assessments go beyond efforts to identify the natural resources at greatest risk due to agricultural production activities and instead provide scientific input for analyses of the cost-effectiveness of the conservation programs.

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A. SUMMARY AND RECOMMENDATIONS

We have reviewed the risk assessments prepared by the U.S. Department of Agriculture (USDA) in support of regulations implementing the Conservation Reserve Program (CRP) and Environmental Quality Incentives Program (EQIP).[*] These two natural resource conservation programs were authorized as part of the 1996 Farm Bill. The risk assessments are required to support an evaluation of these two Programs' cost-effectiveness.

We found that USDA conducted the assessments using methods generally consistent with the current best practices in the field. Nevertheless, improvements in the assessments can be made, particularly in the areas of assessment endpoint selection, definition, and estimation. Many of the assessment endpoints were too diffuse or ill-defined to provide an adequately sharp characterization of the benefits that may be expected from implementation of the two programs.

Two reasons for this lack of clarity were apparent. First, Congress presented the Department with a large, unprioritized set of objectives for the two programs. Frequently, these multiple objectives are conflicting, and at a minimum, they are competing. Further, little appears to be known concerning the public's priorities for environmental improvement and natural resource conservation. These two factors contributed to the use of a disparate set of assessment endpoints. Stakeholder negotiations, however useful and necessary, seem a poor substitute for identifying natural resource values and priorities of the general public, and a locally-led implementation process may fail to target the programs so as to maximize environmental benefits at the national level. Second, for many of the environmental factors affected by agriculture, there does not exist, even in the professional community, broad agreement on what changes should be considered "adverse" and which may be considered negligible. Identification of adverse effects, and thresholds for their action, is not clear, in part because these are heavily value-dependent. In addition, the natural resource scientific and professional community lacks conventional tools and models for linking things that they can measure with precision in the field or lab to many environmental resource values that may be impacted by agricultural activities. This is particularly true when natural variability in conditions is high and when impacts are cumulative over large geographic scales and extended periods of time. We noted, however, that the assessments make only limited use of the empirical data and analyses that are available.

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[*] This report was conducted under a cooperative agreement with the U.S. Department of Agriculture, Office of Risk Assessment and Cost Benefit Analysis. We acknowledge contributions to this review made by Ms. Tonalee Key of Hamilton, N.J.

Finally, we found some "double counting" of program benefits. The CRP and EQIP are, in part, intended to assist agricultural producers in complying with mandatory regulations. The environmental benefits of regulatory compliance would occur, however, absent the programs.

We recommend that in its continuing efforts to conduct regulatory analyses of these major conservation programs, USDA:

1. **Seek clarification for agricultural conservation policy directions and priorities. Endpoints selected for future risk and benefit analyses should reflect and inform these directions and priorities.**

2. **To help clarify policy preferences, consider obtaining sophisticated information on public natural resource values and priorities. This information can be obtained through use of systematic focus groups and public opinion surveys. Because this information is needed by other government agencies, joint projects to obtain it may be cost-effective.**

3. **Keep in mind that the primary purpose for national-level risk assessments should be to provide scientific input for analyses of cost-effectiveness of efforts to mitigate impacts of agriculture on human health and the environment. Since these are constructed by aggregating regional observations and analyses, it may be possible to organize the regional efforts so as to provide support for guidance in interpreting national policies once these are set.**

4. **Consider comparing the estimated environmental benefits of centrally-targeted programs with those resulting from locally-led program implementation.**

5. **Develop the means to identify benefits and costs that can be uniquely attributed to CRP and EQIP.**

6. **Consider adopting habitat diversity as an important and practicable indicator of biodiversity.**

7. **Employ a robust procedure to evaluate expert opinion about what constitutes an adverse health or environmental effect.**

8. **Be more specific about defining risk assessment endpoints and measures and formally elaborate the linkages between field and lab measures and natural resource values.**

B. INTRODUCTION

This paper reports an evaluation of the first steps taken by USDA toward conducting focused analyses of the potential effects of national natural resource conservation programs and makes recommendations for improving the practices of future analyses.

The Federal Crop Insurance Reform and Department of Agriculture Reorganization Act of 1994 (P.L. 103-354, Sec. 304) required the USDA to begin conducting risk assessments of major regulations (those having an annual economic impact of at least $100 million) proposed by the Department the primary purpose of which is to regulate issues of human health, human safety, or the environment. The 1994 USDA Reorganization Act also established the Office of Risk Assessment and Cost-Benefit Analysis to conduct internal review of regulatory analyses.

The Department's first assessments of national natural resource conservation programs were conducted in the process of developing regulations to implement the conservation provisions of the 1996 Farm Bill (Title III of the Federal Agriculture Improvement and Reform Act of 1996, P.L. 104-127, H.R. 2864).[1] Two programs qualified as major regulations, the Conservation Reserve Program (CRP) and the Environmental Quality Incentives Program (EQIP). The Farm Service Agency (FSA) administers the CRP. The Natural Resources Conservation Service (NRCS) manages EQIP. The CRP was established by the 1985 Food Security Act (P.L. 99-198) and redirected by the Food, Agriculture, Conservation, and Trade Act of 1990 (P.L. 101-624). The program pays farmers to retire land from production under 10-year contracts. The 1996 Farm Bill caps land enrollment in the CRP at 36.4 million acres, or roughly the current level of the system. The program's annual budget is somewhat less than $2 billion. The 1996 Farm Bill merged four USDA cost-sharing programs under EQIP: the Agricultural Conservation Program (authorized in 1936), the Great Plains Conservation Program (authorized in 1956), the Colorado River Basin Salinity Control Program (authorized in 1974), and the Water Quality Incentives Program (created by the 1990 Farm Bill). Thus, EQIP will assist agricultural producers in addressing a variety of natural resource conservation problems. The 1996 Farm Bill (Sec. 341) mandates that 50 percent of EQIP funds go to livestock producers. The annual EQIP budget is $200 million (CRS 1997).

In agreement with usage common in this profession, the analyses reviewed here are called "risk assessments." As with many such analyses, these logically form the initial part of an analysis of benefits, in this case benefits flowing to the public from the CRP and EQIP expenditures. They describe the harm to the environment caused by present agricultural practices, harm that is expected to be reduced by these two Programs. Strictly speaking, these and similar analyses might better be termed "benefits assessments." [2] By its own

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[1] See *Fed. Reg.*, Vol. 62, No. 33 (Feb. 19, 1997), pp. 7601-7635 for the final Conservation Reserve Program regulations. See *Fed. Reg.*, Vol. 62, No. 99 (May 22, 1997), pp. 28257-28292 for the final Environmental Quality Incentives Program regulations. The regulations summarize the CRP and EQIP environmental risk assessments and provide USDA contacts for obtaining copies of the complete assessments.

[2] One can construe Congress's intent in requiring "risk assessments" for certain Department proposed regulations to include an analysis of the potential adverse consequences — "risks" — *from* the regulations, as

acknowledgement, however, the Department's risk assessments for CRP and EQIP are incomplete. The EQIP risk assessment (USDA 1997a, p. 6), for example, concedes that it is "not a programmatic risk analysis nor an assessment of the mitigation efforts supported by EQIP and the expected beneficial environmental effects." In reality, USDA simply had insufficient time to plan and conduct a thorough risk assessment. The enormous omnibus Farm Bill was belatedly passed in April 1996, and there was a crush to promulgate the final rules before the 1997 planting season. Therefore, the Department and the Office of Management and Budget (OMB) agreed that these risk assessments could not meaningfully contribute to the Regulatory Impact Analyses (RIAs) required under Executive Order 12866. In issuing the final rules to implement the CRP and EQIP in February and May of 1997, respectively, the USDA announced that it would complete the programmatic benefit-cost analyses based on risk assessments within a year's time.[3]

The 1996 Farm Bill (Sec. 331) directs the Secretary of Agriculture to administer the conservation programs so as to "maximize environmental benefits for each dollar expended." A thorough and accurate assessment of the environmental risks avoided (and caused) by the CRP and EQIP is a necessary precursor to evaluating and maximizing the cost-effectiveness of the programs. To paraphrase Senator Daniel Patrick Moynihan, $2.2 billion a year may not be too much to spend on agricultural conservation programs, but it is too much to spend unwisely.[4]

The evaluation reported here was carried out by conventional peer-review methods: review of the two risk assessments by people skilled and knowledgeable in both risk analysis practices and environmental risk assessment.

C. OBSERVATIONS AND CONCLUSIONS

1. USDA used the best approach now available to evaluate environmental effects from agricultural practices. In conducting the CRP and EQIP risk assessments, the USDA adapted the US Environmental Protection Agency's ecological risk assessment framework (EPA 1992; EPA 1996). The EPA "eco-risk" framework consists of three major components: problem formulation, analysis (consisting of exposure and ecological effects characterization), and risk characterization. Preliminary discussions between risk assessors (i.e., analysts) and risk managers (i.e., decisionmakers, variously defined) are a key element of the framework. During the planning phase of the assessment, agreements are to be reached regarding the intended use and scope of the analysis as well as the selection of "assessment endpoints" and

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well as an analysis of gains from risks reduced — "benefits" — and expected costs associated with realizing those gains. This intent was made explicit in related legislative initiatives of the 103rd and 104th Congresses. In this case, while risks posed by these Programs can be identified (an increase in the price of food, for example), it is not obvious that doing this analysis would add much value. Congress also clearly decided that those risks are small compared to the Programs' expected benefits.

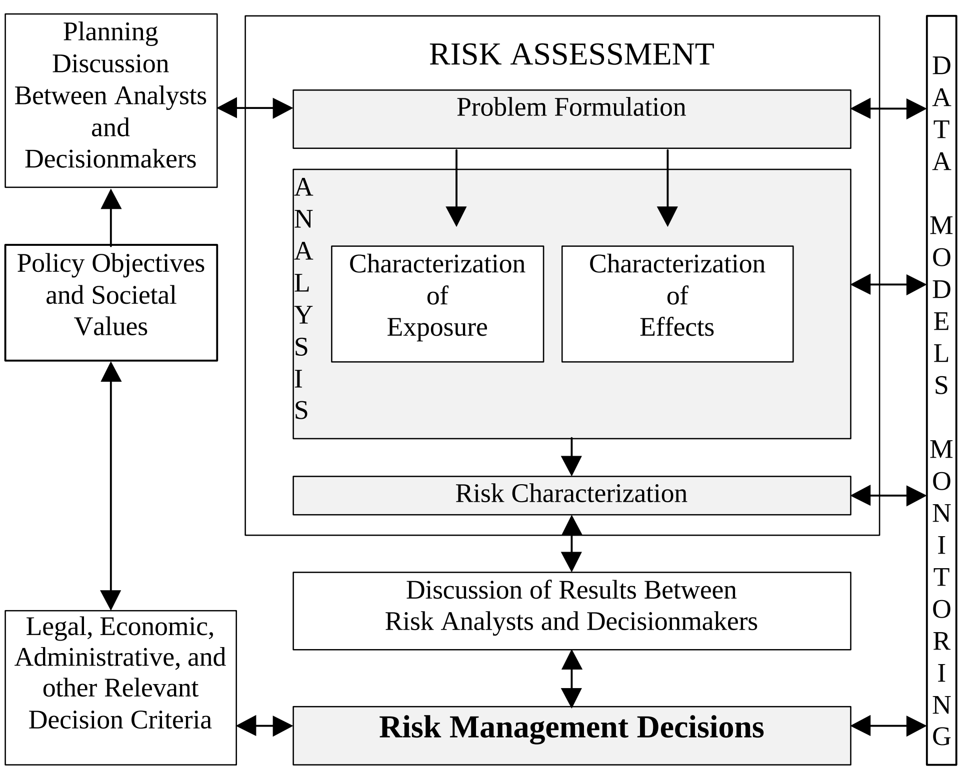
[3] Office of Management and Budget review delayed the final EQIP rulemaking.

[4] Sen. Moynihan made a similar remark about federal pollution control programs while proposing the Environmental Risk Reduction Act of 1993.

measures of effect. Definition of risk management goals for the assessment should center on what decision the assessment is intended to inform and what level of analysis is commensurate with the decision. Assessment endpoints are health, safety, or environmental attributes that are both scientifically relevant and reflect policy goals and societal values. Measures of effect (also called "measurement endpoints," for example, trout mortality) are used to evaluate the response of assessment endpoints (e.g., maintenance of wild fish populations) to environmental stressors (e.g., nitrate levels in a stream that lead to increased algal growth and subsequent oxygen depletion).

Risk characterization provides the basis for communications between analysts and decisionmakers regarding the risk assessment results. The risk characterization combines the results of exposure and environmental effects analyses to evaluate the likelihood, magnitude, severity, spatial and temporal distribution, and reversibility of adverse effects on human health, safety, and the environment. It should discuss and evaluate the strengths and limitations of the analyses, the assumptions used and their effects on the results, and the principal scientific uncertainties. The results of the risk assessment are considered in combination with statutory mandates, economic analyses, administrative factors, evaluation of program performance, and other relevant criteria in risk management decisionmaking. Figure 1 displays the framework for risk assessment and places it in the larger context of risk management decisionmaking.

Figure 1. Framework for Risk Assessment and Risk Management



Adapted from EPA (1992).

As noted above, we found that the EQIP and CRP risk assessments are appropriately considered as works-in-progress. We observed that the two assessments made some efforts to identify natural resources at greatest risk from agricultural activities at regional scales. We conclude that because of the way input data are obtained, stored, and analyzed, the national-scale assessments could logically be constructed by aggregating observations and analyses done at a regional, or even local, scale. Furthermore, if future regional-scale analyses were conducted before initiating the national-scale assessments, such analyses might have independent utility to inform regional program implementation as well as national policies and priorities.

2. It appears that efforts to target CRP/EQIP expenditures so as to maximize environmental benefits were constrained by several conflicting policy decisions. It is readily apparent that the primary administrative objective of the EQIP and CRP risk assessments (USDA 1997a; USDA 1997b, respectively) was to identify the natural resources most "at risk" from agricultural production activities in order to target the programs at priority environmental problems. As such, the risk assessments attempt to respond to decisionmakers' needs. In the debate preceding passage of the 1996 Farm Bill, there were many demands emanating from policymakers, the scientific community, and the public to better "target" the disparate, decentralized USDA conservation programs. The National Research Council Committee on Long-Range Soil and Water Conservation, for example, strongly emphasized the importance of targeting the agricultural conservation programs at those regions where environmental degradation is most severe and at "problem farms" that cause a disproportionate share of environmental problems. The Committee concluded, "The inability or unwillingness to target policies ... at problem areas and problem farms is a major obstacle to preventing soil degradation and water pollution" (NRC 1993). Similarly, the World Resources Institute recommended targeting agricultural conservation programs to areas where soil and water quality problems are worst and to other environmentally sensitive areas (WRI 1995). The grassroots sense that the conservation programs need to be better targeted to "achieve more bang for the buck" emerged from a series of regional forums and focus groups on agriculture and the environment (SWCS 1995).[5]

It appears, however, that the Department's efforts to target the conservation programs are severely constrained by conflicting demands to distribute program expenditures geographically, to limit potentially adverse effects on agriculturally-dependent local economies, and to satisfy an increasing set of environmental objectives. Others question whether the strategy of targeting

[5] Ribaudo (1986) suggested that the off-site impacts of soil erosion should be considered in targeting regions for soil conservation programs. Regions where a reduction in erosion on agricultural land would improve water quality are identified and compared to regions targeted using on-site criteria. Differences in the regions identified indicate that failure to consider both onsite and offsite impacts may result in inefficient targeting. Ribaudo et al. (1994) concluded that crop land retirement as a primary water pollution control tool is expensive, but if appropriately targeted, could generate sufficient benefits to outweigh social costs.

assistance at problem farms rewards bad actors (see SCWS 1995) or whether voluntary, "best management practice" programs will sufficiently mitigate environmental problems arising from problem farms or affect the environmental performance of the most recalcitrant producers (see NRC 1993). Below is a brief discussion of some of the constraints that complicate or thwart efforts to target the conservation programs. However, the bottom line is that the USDA risk assessments' efforts to target don't.

In addition, while a coarse-grained targeting serves the necessary function of bounding the scope of a doable analysis, this constitutes only the first step of a thorough risk assessment appropriately designed to provide input for an economic regulatory impact analysis.

Targeting bounds the scope of a doable analysis, but it is only the first step toward a regulatory impact analysis.

In response to calls for targeting conservation programs, the 1996 Farm Bill (Sec. 331) authorizes the Secretary of Agriculture to designate regions or watersheds as Conservation Priority Areas. While the Senate version of the bill specified that the Conservation Priority Areas include the Chesapeake Bay Region, Great Lakes Region, Rainwater Basin Region, Lake Champlain Basin, the Prairie Pothole Region, and Long Island Sound Region (S. 1541, Sec. 311), the final conference bill designated no priority areas and directed the Secretary to use the services of local, county, and state committees in carrying out the conservation programs. Policymakers had agreed that instead of a top-down priority-setting process, state and local levels would have a strong role, primarily through State Technical Committees (CRS 1997), whose memberships were expanded beyond government representatives to include conservation experts and stakeholders (H.R. 2854, Sec. 342). In promulgating the final regulations for the CRP, the Secretary designated the Chesapeake Bay Region, Long Island Sound, Great Lakes Region, and Prairie Pothole Region as Conservation Priority Areas. State FSA committees also may submit recommendations regarding other CRP priority areas. Regions are eligible for priority area designation if the region has significant water quality of wildlife habitat problems related to agricultural production or if the designation helps agricultural producers to comply with federal and state environmental laws (Sec. 1410.8). Given that agriculture is a principal source of impairment to surface water quality in the U.S. (USDA 1997a, Table 3.2.2.b), however, the criteria fail to narrow very much the list of potential priority areas. Under EQIP rules, the NRCS Chief may designate "national" Conservation Priority Areas, whereas the NRCS State Conservationist approves locally recommended priority area designations and makes funding recommendations for program activities in the state.

Our reading of the CRP and EQIP risk assessments is that while the documents contain much information that could potentially guide an effort to target the programs regionally (at least by objective), the assessments seem to make every effort not to limit the options of decisionmakers. The EQIP risk assessment, for example, uses data from the 1992 NRCS National Resources Inventory to determine that excessive soil erosion is concentrated in:

- the lower Mississippi River Basin
- the Ohio River Basin

- the Tennessee and Cumberland River Basins
- portions of the Missouri Valley Basin, including Iowa and northern Missouri, and
- the western Great Plains areas subject to high wind erosion (USDA 1997a, p. 79).

In discussing EQIP priorities on a regional basis, however, soil erosion is found to have a significant cumulative impact in every region (USDA 1997a, pp. 127-151). The EQIP assessment also presents state water quality data compiled by EPA showing that two regions, the Corn Belt and Delta States, account for approximately 60 percent of the nation's rivers and streams impaired by siltation and nutrients (USDA 1997a, p. 97). Nevertheless, sedimentation/siltation is found to have a significant cumulative impact in 9 out of 10 regions (the Mountain Region being the exception), and nutrient runoff/leaching is found to be significant in 8 out of 10 regions (excepting the Mountain and Pacific Regions) (USDA 1997a, pp. 127-151). Similarly inclusive is the CRP discussion of agricultural impacts on water quality in 47 states (USDA 1997b, pp. 62-70) and the list of wildlife habitat priority areas: Cornbelt, Prairie Pothole Region, Southern Plains, Lower Mississippi Valley, Platte River Headwaters, Great Lakes Basin, Endangered Species Habitats (occurring in every state), National Grasslands, habitat associated with Federal- and State-managed wildlife and recreation areas, and croplands in riparian zones (USDA 1997b, pp. 98-105).

The 1996 Farm Bill contains some provisions that preclude an unconstrained targeting of the conservation programs across regions. The requirement that 50 percent of EQIP funds be available to livestock producers, for instance, unmistakably limits the opportunity to target cropland areas. Another constraint is the prohibition, motivated by concern about local economic impacts, against enrolling more than 25 percent of the cropland in any county in the conservation programs (H.R. 2854, Sec. 1243). (In the past, the 25 percent county ceiling has limited participation in CRP in some areas, especially in the Northern Plains (CRS 1994).)

Another dynamic that confounds targeting efforts is the growing list of environmental objectives for the conservation programs. Historically, agricultural conservation programs were designed to address natural resource problems on the farm, primarily soil erosion and inefficient water use. Traditionally, USDA conservation programs emphasized sustaining and enhancing long-term agricultural productivity. Sedimentation of water bodies, which is closely associated with erosion, is the traditional off-site damage of concern due to effects on flood control, navigation, power generation, drinking water treatment, and fisheries. Over time, however, the number of conservation program objectives has grown considerably. The Colorado River Salinity Control Program was established in 1974 and intended to improve irrigation practices to help meet treaty obligations with Mexico. In the last three Farm Bills, Congress has broadened the agricultural conservation agenda by explicitly including water quality, wetlands, and wildlife habitat to the list of program objectives. The recent EQIP and CRP risk assessments reveal that the Department has expanded the set of conservation objectives beyond those explicitly mentioned by Congress to include targets such as threatened and endangered species, air quality, open space, the visual landscape, and cultural and historic resources. Specifying a single or dominant objective such as erosion control would make the program easier to target and administer but takes a narrow view of environmental damages associated with agricultural practices. *Recognizing multiple program objectives reflects a more robust vision of environmental quality, but the numerous, unranked objectives seem unwieldy and may invite* ad hoc *decisions arrived at through bargaining.*

3. The selection of assessment endpoints seems unfocused and somewhat arbitrary, and may not adequately reflect public values. The assessment endpoints selected for a risk analysis should derive from statutory and administrative policy objectives and societal values, concerns, and priorities. The CRP assessment reported that these endpoints were used (USDA 1997b, p. 4):

- soil productivity (or quality),
- surface water and groundwater quality,
- wetland function and values,
- wildlife habitat,
- air quality.

For the EQIP assessment, these were said to have been used (USDA 1997a, p. 77):

- structure of off-site resources and habitats,
- livestock or plant yields,
- wetland and riparian functions,
- viability of aquatic communities,
- survival of threatened and endangered species,
- good air quality,
- diversity and extent of natural habitats,
- quality of cultural and historic resources,
- potable water supplies,
- diversity of terrestrial and avian wildlife species,
- quality of landscape resources.

For risk policy observers unfamiliar with agricultural conservation programs, the USDA's selection of agricultural yields as a risk assessment endpoint may raise questions. The considerable emphasis placed on the on-site effects of soil erosion seems particularly peculiar given projections that agricultural yield losses resulting from erosion would be less than 10 percent over the next 100 years (NRC 1993, p. 193). It also may be legitimate to question why taxpayers should subsidize agricultural producers for not depleting or degrading their privately owned natural capital. The 1996 Farm Bill, however, states that, in general, the conservation programs are intended "to conserve and enhance soil, water, and related natural resources, including grazing land, wetland, and wildlife habitat" (Sec. 331). EQIP is specifically intended to assist "farmers and ranchers that face the most serious threats to soil, water, and related natural resources" (Sec. 334). EQIP and the technical assistance and cost-sharing components of CRP are intended to "reconcile productivity and profitability with protection and enhancement of the environment" (Sec. 331). These provisions likely derive from the policy goals of sustainable agriculture, food security, and supporting the farming enterprise. Therefore, it is evident that the selection of risk assessment endpoints associated with the productivity of on-site resources conforms to statutory guidance.

An indirect connection also can be found between statutory guidance and the selection of air quality as an assessment endpoint. The 1996 Farm Bill (Sec. 331) states that Conservation Priority Areas may be designated to assist agricultural producers to comply with . . . Federal and State environmental laws, and a statutory objective of EQIP is to assist “farmers and ranchers in complying with . . . Federal and State environmental laws” (Sec. 334). Agricultural practices (e.g., tillage and burning to clear land or eliminate crop residue) may contribute to the non-attainment of National Ambient Air Quality Standards for Particulate Matter in some regions. The risk assessments also define the air quality assessment endpoint to encompass other dimensions that arguably address societal values or concerns—odor, visibility, acidic deposition, and greenhouse warming potential.

It is fair to describe these air quality values and the cultural, historic, and landscape resource values selected by USDA as risk assessment endpoints as falling under the heading of “other natural resources” related to those specified by Congress (i.e., soil, water, grazing land, wetland, and wildlife habitat). USDA certainly has the authority and obligation to exercise reasonable administrative judgement and discretion in assessment endpoint selection.

However, the conservation program assessments do not explicitly incorporate or respond to any studied effort to elicit public input regarding values or priorities for the conservation programs.[6] (The selection of assessment endpoints may have been consciously informed by exercises such as the forums and focus groups reported in SWCS (1995), but, if so, no reference to such exercises is included.) For example, given the growing concern about the evolution of drug-resistant bacteria, it is plausible that members of the public may have identified the development of resistant strains from the use of antibiotics on livestock (e.g., to limit economic losses from *Staphylococcal mastitis* in dairy operations) as a public health issue that EQIP could address. This particular example may or may not have been included for consideration as a risk assessment endpoint for the conservation programs. Nevertheless, it is not outwardly apparent that the assessments benefited from an informed public discussion. *Public input to priority-setting seems essential for maximizing program cost-effectiveness. In addition to convening stakeholders, input should be obtained systematically by conducting focus groups and public opinion surveys designed to reflect the preferences of the general public.*

4. The assessments are mixed in terms of providing well-defined assessment endpoints and establishing links between assessment endpoints and measures of effect. Some of the endpoints chosen are clearly focused, for example, livestock and plant yields and survival of threatened and endangered species (given the finite number of officially classified species). Although soil quality depends on the soil’s capacity to provide a number of ecological functions, the assessments emphasize productivity, which functionally relates to

[6] The regulations implementing the conservation programs were, of course, subject to the obligatory public notice and comment process. Risk assessments, however, are intended to be input to regulatory proposals.

promoting plant growth.[7] Water quality is reasonably well defined as meeting designated uses, i.e., fishable, swimmable, potable, and suitable for irrigation.

However, wetlands are vaguely defined in the assessments (e.g., swamps, floodplain forests, bogs, marshes, and prairie potholes). Presumably wetlands determinations would actually be made in accordance with the 1994 Memorandum of Agreement between the USDA, EPA, the Department of the Interior, and the Department of the Army to follow the 1987 Corps of Engineers Wetland Delineation Manual (as modified and clarified by the Corps in 1991 and 1992) for delineating wetlands on areas where the native vegetation is intact and to use the 1994 NRCS National Food Security Act Manual for delineating wetlands on areas where the native vegetation has been removed due to ongoing agricultural activities. Wetland functions are discussed (e.g., water purification; flood and storm protection; provision of habitats for fish, wildlife, and threatened and endangered species; and supply of hunting and recreational opportunities), but these functions are not operationally defined. The water quality benefits of wetlands could be more explicitly defined, for example, in terms of sequestering and/or degrading specific contaminants (e.g., suspended solids and nutrients).

Other assessment endpoints are altogether vague—"viability of aquatic communities," for example. Commonly, viability refers to a specific population or species, and an appropriately defined assessment endpoint would be, for example, Coho salmon breeding success and fry survival in the Elk River watershed. But it can be unclear what is meant by the "viability" of an assemblage of co-occurring populations. In order to clarify the meaning of the assessment endpoint, community viability may be linked to an explicit program objective—e.g., protect X% of the individuals in Y% of the species in aquatic communities—and can be expressed in terms of the reproduction and survival of the species. An appropriate assessment endpoint for the ill defined "wildlife habitat" might be change in critical (e.g., breeding) habitat structure. The conservation program assessments leave "diversity of natural habitats" and "cultural and historic resources" undefined as well. (A methodological approach for measuring and targeting diversity in a habitat conservation network is proposed in the Recommendations section below.)

The conservation program risk assessments lack thorough discussion of measures of effect or how these will be linked to risk assessment endpoints. Certainly, there are conventional measures of plant and livestock yields. And as specified by the Department (*Fed. Reg.*, Vol. 61, No. 108, June 4, 1996), soil loss due to sheet and rill erosion will be estimated utilizing the universal soil loss equation (USLE) and the revised universal soil loss equation (RUSLE), and soil loss due to wind by the wind erosion equation (WEQ). Erosion estimates are linked to soil productivity or quality through the soil loss tolerance (T-value), which is the estimated maximum annual rate of annual soil erosion that will permit crop productivity to be sustained economically and indefinitely. On the other hand, the

[7] Soil quality is conventionally defined in terms of plant growth promotion, but soil performs other ecological functions such as regulating and partitioning infiltration and runoff and buffering environmental pollutants. Currently, however, there appear to be no conventionally accepted indicators for the non-productivity aspects of soil quality (NRC 1993).

assessments crudely measure the human health and ecological effects of pesticides by bulk application rates (tons/yr.), which ignores differences among pesticides in toxicity, environmental persistence, and bioaccumulation potential. There is furthermore no defined or referenced set of algorithms linking estimated on-site erosion rates to off-site sedimentation rates that could be used in turn to estimate the need to dredge rivers to maintain navigable channels, the extent of wetland habitats, or exceedances of ambient water quality standards.

In addition to the severe time and resource constraints under which the conservation program risk assessment teams were operating, the failure to specify how the programs' impact on assessment endpoints is to be estimated may derive from unease about the large magnitude of the associated uncertainties. In what seems to represent the consensus judgement of soil and water scientists, NRC (1993) states that the current understanding of the effects of farming systems on soil and water quality is generally sufficient to identify best management practices but is not sufficient for making quantitative estimates of how much environmental improvement will result from implementing alternative methods. What this statement more accurately reflects, however, is discomfiture *among the scientists* with the uncertainties surrounding such estimates. The conservation program assessments nevertheless cite—without further elaboration—economic studies (Clark et al., 1985; Colacicco et al., 1989; Ribaudo et al., 1989) providing quantitative estimates of on- and off-site damages from agricultural erosion and sedimentation and the reduction in those damages associated with existing conservation programs. Of necessity, such studies require analysts to link those things that can be measured with some precision to valued environmental resources. The risk assessments therefore implicitly adopt the data, methods, models, and assumptions used in these studies without fully and explicitly discussing and evaluating them.

5. Identification of adverse effects, and thresholds for their action, is not clear, in part because these are heavily value-dependent. Closely related to the selection of risk assessment endpoints is the issue of determining what constitutes an "adverse" human health or environmental effect. This can be particularly difficult for ecological assessment endpoints. Traditionally, for example, the only "ecological" effects of concern related to pesticides have been glaring bird and fish kill episodes. A fish or bird kill episode that some members of society find offensive may have inconsequential long-term impacts on the local population, however, due either to a local reproductive response or to immigration from adjoining populations. On the other hand, such an event may result in tangible welfare impacts (due to disposal costs or lost sales for businesses such as lakeside food stands that rely on a pleasant landscape).

When an environmental effect varies in intensity or extent, defining what constitutes an adverse effect generally requires decisionmakers to define a threshold—if any—below which effects are considered negligible. In some cases, the conservation program assessments contain multiple, inconsistent thresholds. For other resource values, the assessments make no determination of what magnitude of change constitutes an adverse impact.

The CRP assessment suggests three distinct thresholds for soil erosion: 1) land with unsustainable erosion (T, as defined above); 2) highly erodible land (HEL, defined as land with an erodibility index (EI) above 8. EI is the factor by which potential erosion exceeds the soil loss tolerance); and 3) land with projected yield loss greater than 2 percent in 100 years. The difference among threshold indicators has regional implications. The T and HEL thresholds suggest that erosion problems are concentrated in the Corn Belt, Northern Plains, and Mountain Production Regions, while the productivity loss threshold suggests that problems are greatest in the Corn Belt and Lake States Regions (USDA 1997b, Table 3.1). (According to NRC (1993), T may not be optimal, but it is the best available threshold indicator for soil quality.) The EQIP assessment states that in the Northeast, the program should be directed toward areas of soil loss of 2T or more, whereas in the Delta States, Corn Belt, and Lake States, the threshold of concern appears to be as low as 1T (USDA 1997a, pp. 130-140).

For some endpoints, the risk assessments are not explicit about what is an adverse effect, but there are ready-made benchmarks. Ambient air and water quality standards provide thresholds for air and water quality assessment endpoints. Maximum Contaminant Level Goals for drinking water provide thresholds for the assessment endpoint of potable water.[8] Similarly, there are recognized soil water alkalinity/acidity and salinity levels that lead to substantial yield reductions for particular crops. In other instances, however, there are few guideposts. The EQIP team, for example, defines the “off-site resources and habitats” assessment endpoint in terms of changes in the structure of aquatic and terrestrial habitats resulting from increased severity or frequency of flood events. Implicitly, therefore, the assessment does not necessarily regard changes in off-site species composition or relative abundance as ecologically adverse. Instead, ecological significance is associated with a more fundamental change in the ecosystem, e.g., from a previously terrestrial habitat to a permanent water impoundment. Over time, however, there are natural shifts in a stream course. What magnitude of change in the structure of off-site resources and habitats will USDA decisionmakers consider to be an adverse effect resulting from flood events? Similar questions arise about other assessment endpoints. What constitutes an adverse impact on habitat diversity or the extent of wetlands? Need changes in community species composition be irreversible over the typical human life-span to be regarded as adverse?

EPA (1996, p. 146) proposes five criteria for evaluating adverse changes in assessment endpoints:

- Nature of effects
- Intensity of effects
- Spatial scale
- Temporal scale
- Potential for recovery

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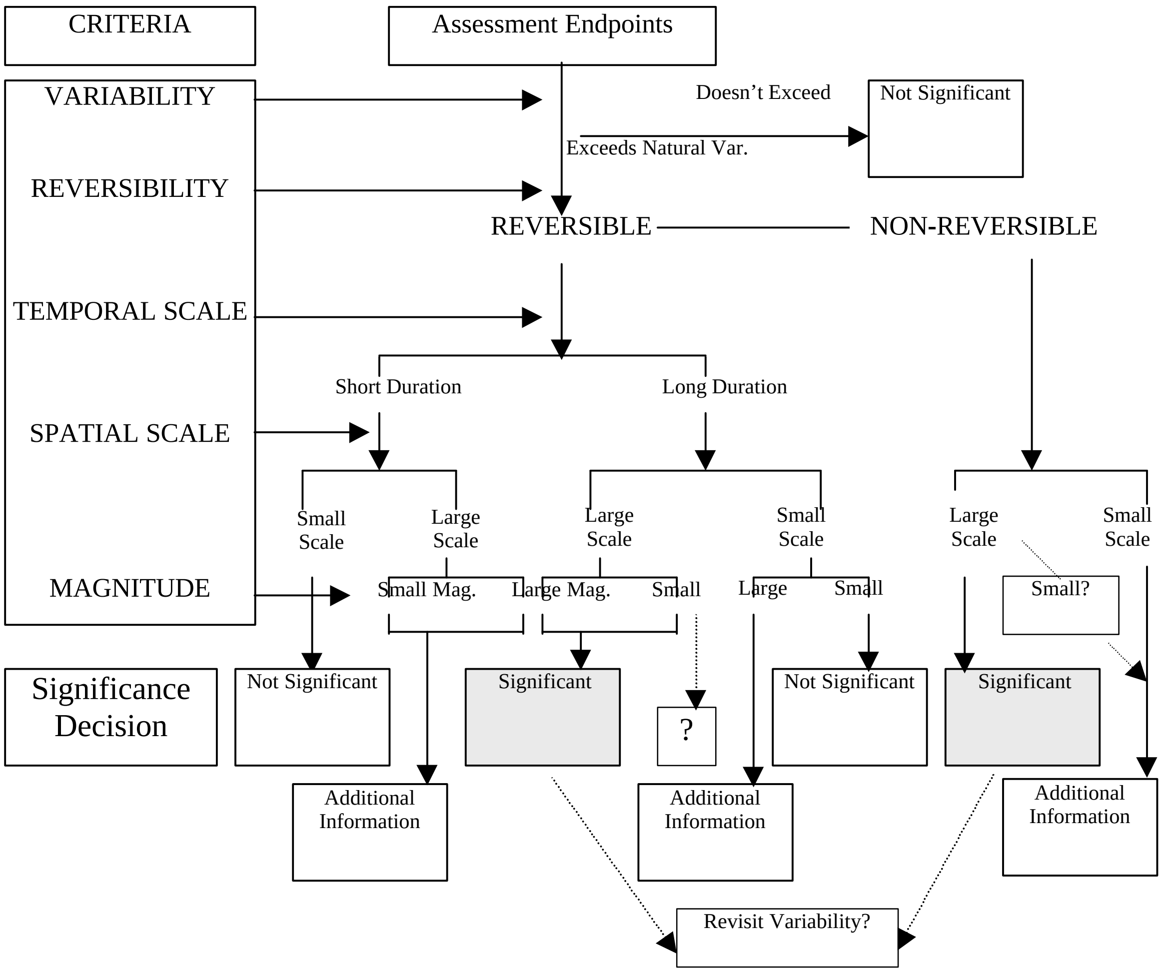
[8] Enforceable drinking water standards do not necessarily constitute a no effects threshold for potable water because National Primary Drinking Water Regulations take into account technical feasibility and affordability, primarily from the point of view of large public drinking water suppliers.

One of the thorniest science policy issues in risk analysis is whether observed or predicted changes in assessment endpoints are distinguishable from the natural variability. This applies to human health risk assessment (for example, what level of temporary lung function impairment is considered “adverse”) but is particularly true in the case of ecological assessment endpoints. Stream flows, animal populations, and the like are subject to large natural fluctuations. Another complicating factor is that very limited time-series data are available for many ecological endpoints of concern. As a result, statistical tests often have very weak power to detect effects that may otherwise be regarded as substantial.

Harwell et al. (1994) provide ambiguous—but realistic—guidance on determining the significance of ecological changes. For an ecological change to be judged significant, according to Harwell and colleagues, it must exceed some estimate of natural variability, but statistical significance need not be demonstrated. Another way of phrasing this is simply to say that the 95 percent statistical confidence interval is not divinely ordained. In the regulatory context, the determination of an allowable rate of false negatives/positives is a policy judgement that should be informed by science. Risk managers may choose to follow scientific conventions for the sake of consistency or to avoid *ad hoc* decisionmaking, but there is no “scientifically correct” or purely “objective” answer. (An approach for eliciting and characterizing expert opinion to inform some policy decisions about thresholds when the effect varies in intensity/magnitude is proposed in the Recommendations section below and further elaborated in the Appendix.)

An analysis of the “Ecological Significance Framework” proposed by Harwell et al. (1994) (Figure 2) underscores that subjective judgements are inherent in making a determination about the significance of an environmental change. It is not self-evident, for example, why a reversible effect of long duration, large scale, and small magnitude should necessarily be considered a significant change. An example of this sort would be a small, long-term increase in global surface temperatures, the significance of which is currently being hotly debated in the policy domain. (Alternatives to Harwell’s framework are indicated in Figure 2 by dashed lines.) Similarly, it is not readily apparent why the magnitude of the effect should not be taken into consideration for non-reversible, large-scale changes. If such a change were of relatively small magnitude, a decisionmaker might choose instead to acquire additional information before making a final determination. Harwell and colleague’s framework reemphasizes that discerning whether a change exceeds some estimate of natural variability is critical. Harwell et al. (1994) suggest that a default position is to assume that exceedance of natural variability is the norm rather than the exception; then based on that assumption, one may proceed through the ecological significance decision tree and reassess variability only if it is deemed to be the critical component in the decision. An alternative default, however, would be to consider any determination of ecological significance to be provisional subject to a reassessment of variability. This discussion is not intended to criticize Harwell’s proposed framework but rather to emphasize the degree to which risk assessors and risk managers must exercise judgement in determining what constitutes an adverse ecological effect. As suggested above, this decision may be scientifically elaborate, but it is ultimately a policy call.

Figure 2. Harwell's Framework for Ecological Significance.



Adapted from Harwell et al. (1994)

6. The assessments make only limited use of the empirical data and analyses that are available. Much of the data and analysis available for risk assessment of agricultural conservation programs is highly disaggregated, with much information available only at the plot-, field-, or county-level. Considerable regional- and national-level data and analysis are available, however. The USDA assessments make limited use of the Third Resource Conservation Act Appraisal (RCA III) prepared by NRCS, which focuses on the nexus between agriculture and water quality. The CRP and NRCS assessments make no reference to other relevant large-scale studies such as those conducted by the Center for Agricultural and Rural Development (CARD) at Iowa State University with support from EPA. As indicated above, part of the problem may be that science analysts in the agricultural field are uncomfortable with the large uncertainties inherent in regional- and national-scale analyses. Some may feel that it is better to provide policymakers with no analysis than to provide them with highly uncertain analysis. This simply begs the question, better for whom? National-level analysis of the cost-effectiveness of the conservation programs is not elective—better that it be based on the best available scientific analysis and be explicit about the magnitude and effects of the underlying uncertainties.

7. Some "double counting" of benefits probably occurs. A fundamental problem with the EQIP and CRP risk assessments has to do with their scope. They fail to distinguish public health and environmental risks that are *uniquely* avoided by the programs from the results of risk management that would occur in the absence of the voluntary programs as agricultural producers comply with mandatory federal, state, and local regulations. To some extent, the statutory purpose of the agricultural conservation programs may be liable for the fact that the assessments capture risks that would otherwise be reduced.

For example, the 1996 Farm Bill states that Conservation Priority Areas may be designated to assist agricultural producers to comply with nonpoint source pollution requirements under the Clean Water Act and other Federal and State environmental laws (Sec. 331). EQIP is explicitly intended to assist "farmers and ranchers in complying with . . . Federal and State environmental laws" (Sec. 334). (Note that this objective is inconsistent with the statutory requirement—also included in Sec. 334—to maximize environmental benefits per dollar expended.) Congress also implicitly directed USDA to subsidize compliance of some livestock producers that are subject to mandatory EPA regulations as point sources under the Clean Water Act. It did so by earmarking fifty percent of EQIP funds for livestock producers and removing the definition of large confined livestock operations from the Senate version of the Farm Bill (S. 1541) that are ineligible for EQIP. (The final version of the 1996 Farm Bill (Sec. 334) left definition of large confined livestock operations to the Secretary. Under the final EQIP regulations, the NRCS State conservationist may develop a definition for a large confined livestock operation as it applies to that particular State using criteria recommended by the State technical committee (Sec. 1466.7).) It is clearly within the legitimate authority of public policymakers to choose to subsidize regulatory compliance by any particular economic sector. It is also plain, however, that including the environmental benefits of mandatory programs—whether they be federal, state, or local—within the scope of risk assessments for voluntary programs results in a double-counting of risk reductions.

We note, however, that the Department currently lacks the means to identify benefits reliably assignable to these broad programs, so it is impossible to identify the magnitude of this "double counting."

D. RECOMMENDATIONS

1. Seek clarification for agricultural conservation policy directions and priorities. Endpoints selected for future risk and benefit analyses should reflect and inform these

directions and priorities. We noted that the 1996 Farm Bill directed USDA to pursue conflicting and competing objectives for CRP and EQIP. The legislative history of the Farm Bill makes it clear that Congress was well aware of these built-in tensions that it left for the Department to sort out—subject, of course, to consultation with the relevant oversight committees. It may be worth informing Congress as to the effect of these conflicting directives, with the intent of obtaining clarification of goals and objectives. In particular, Members of Congress may value information about how the public ranks competing conservation program objectives and about the extent to which subsidizing regulatory compliance by agricultural producers impedes the goal of maximizing environmental benefits per dollar expended.

We noted the difficulties associated with identifying the most relevant and useful endpoints for analysis, and suggest some things for consideration below. In general, these endpoints should flow from the policy goals and priorities, so as to characterize risks in a way most useful for decision-makers.

2. To help clarify policy preferences, consider obtaining sophisticated information on public natural resource values and priorities. This information can be obtained through use of systematic focus groups and public opinion surveys. Because this information is needed by other government agencies, joint projects to obtain it may be cost-effective. As analysis and evaluation of the conservation programs continues, USDA should take additional steps to elicit input regarding the natural resources which the public values and public priorities among conservation objectives. Although stakeholder and expert participation may promote effective program implementation, we would recommend in particular that the Department look beyond groups of stakeholders and conservation experts to assess natural resource valuation by the general public. Conducting systematic focus groups and public opinion surveys may be useful in identifying national- or regional-level program priorities. It would be particularly informative to evaluate whether or how the broad-scale natural resource priorities of the general public differ from those revealed by locally-led program implementation, which aggregates the priorities of local stakeholders.

We would caution USDA, however, to be extremely skeptical about the potential to forge a meaningful consensus from a diverse public with conflicting or competing values and to be very cautious about abdicating regulatory decisionmaking authority to interested and affected parties. Public involvement needs to inform risk assessment problem formulation, and a strong consensus on agricultural conservation goals makes it more likely that risk management decisions will be supported during program implementation. Public input, however, is no substitute for making tough policy choices. Priority-setting in particular can become extremely difficult as the number of parties who feel they have a legitimate claim to program resources increases.

Many regard the facilitation of conflict resolution, consensus building, and the sense of shared ownership that comes with participating in problem-solving as essential components in seeking solutions to environmental and natural resource policy problems. Unfortunately, as game theory suggests, win-win solutions are not possible in all cases. As Fisher and Ury (1981, p. 84) observe:

> However well you understand the interests of the other side, however ingeniously you invent ways of reconciling interests, however highly you value an ongoing relationship, you will almost always face the harsh reality of interests that conflict. No talk of "win-win" strategies can conceal that fact.

This information on values is important not only to USDA but also to the Departments of Interior and Commerce. Some of it will also be of interest to EPA. (It may well also be of interest to some producer groups.) Since obtaining reliable information on these values requires sophisticated techniques, it will be expensive to obtain (greater than $1 million). Thus, it might be cost-effective for USDA to join with partners to do the studies needed.

3. Keep in mind that the primary purpose for national-level risk assessments should be to provide scientific input for analyses of cost-effectiveness of efforts to mitigate impacts of agriculture on human health and the environment. Since these are constructed by aggregating regional observations and analyses, it may be possible to organize the regional efforts so as to provide support for guidance in interpreting national policies once these are set. In principle, a prospective risk assessment of the kind reviewed here is done to provide input to the benefits analysis required for a Regulatory Impact Analysis (RIA). These assessments can be expected to provide some input to the RIAs yet to be done. They also can be expected to provide input to Departmental analysis of these and related programs, as part of ongoing performance improvement efforts. Although these programs pose some unique challenges to risk analysts, it remains appropriate for the approach used in the subject assessments to be used for similar future analyses.

In a risk assessment framework, the analysis should cover risks that may result from the program—not just risks avoided by the program—as well as substitution risks. However, because USDA is required to conduct an Environmental Impact Assessment (EIA) to determine whether the conservation programs have potential adverse environmental effects, it is sensible not to duplicate those analyses in the risk assessments. The scope of EIAs, however, is limited to environmental impacts. Consequently, an EIA will not capture other risks that may be associated with a conservation program (e.g., increased food prices that may exacerbate nutritional deficiencies suffered by poor consumers, though we doubt that the conservation programs constitute such a risk.) In our judgement, it is a good practice to have separate teams conduct the EIA and the risk assessment for a program. Doing so maintains some institutional separation between the risk and benefit assessment functions. To some extent, assessments do acknowledge tradeoffs among risks (e.g., reduced tillage resulting in increased herbicide applications), but there is no explicit risk-risk analysis. There is no evaluation, for example, of the net effect of replacing early-outs with newly enrolled lands under the CRP program. Although it may not be feasible to conduct an exhaustive assessment of substitution risks, policymakers should be made aware of what is known about the net effect of major programmatic changes.

National-scale risk assessments cannot be expected to directly guide regional or local program implementation. They can help identify, however, where natural resource and human health benefits are greatest across regions. They also permit cost-effectiveness comparisons across programs. To the extent that people from regional offices are involved in developing national-scale assessments, the knowledge and experience they gain may be useful in targeting regional program implementation. As we noted above, the regional and local components of national-scale assessments could include analyses of threats posed to natural resources by agriculture in the separate regions. Such analyses could serve as raw material for national-scale assessments, and could also provide input to regional implementation guidance, once national policies and priorities were established.

4. Consider comparing the estimated environmental benefits of centrally-targeted programs with those resulting from locally-led program implementation. Targeting of CRP/EQIP resources formed an important part of these assessments. While this targeting is a necessary initial step to limit the scope of the assessment, it is not sufficient to identify resources most at risk. Eligibility and other factors influence the ultimate participation in the voluntary programs, thus introducing considerable uncertainty into the predictions of actual gains from implementation of the programs. The Department may want to consider conducting a prospective risk assessment, to produce a tally of the benefits of a centrally targeted program to provide a benchmark. For example, a prospective analysis might evaluate the benefits that would accrue from the CRP if eligible lands were enrolled so as to maximize the Environmental Benefits Index (EBI) summed over all lands in the reserve system.[9] Subsequent evaluation could then compare the projected benefits with those resulting from locally-led program implementation. Due to the difficulty in quantifying many of the program benefits, local decisionmakers could be held accountable to document their use of scientific and economic criteria rather than having to meet an artificially precise target or quota of program benefits.

5. Develop the means to identify benefits and costs that can be uniquely attributed to CRP and EQIP. For the purpose of providing scientific input for Regulatory Impact Analyses, the risk assessments should characterize the environmental benefits uniquely attributable to the conservation programs. In its ongoing assessments of CRP and EQIP scheduled to be completed during 1998, USDA should give priority to distinguishing those health and environmental benefits which derive uniquely from the conservation programs

[9] USDA uses the EBI to conduct a national review of bids for enrollment in the CRP. Heimlich and Osborn (1993) estimated the environmental benefits of a simulated CRP enrollment from the entire cropland base represented in the National Resources Inventory on the basis of maximizing the EBI per dollar of rental cost at different levels of total program outlay. Ahrens (1993) provides another example of an attempt to define optimal participation in the CRP.

from those which, although they may be supported by the voluntary programs, would otherwise be attained in order to comply with mandatory federal, state, and local regulations.

6. Consider adopting habitat diversity as an important and practicable indicator of biodiversity. The USDA should investigate the utility of using habitat diversity as an acceptable surrogate for biodiversity.[10] We recommend this for four reasons. First, it seems virtually certain that information on all species (including nonvertebrates, soil microorganisms, etc.) occurring in habitats that may be included a protected area network will remain highly incomplete for the foreseeable future—if not forever. Second, to some extent, the concept of species is a useful but artificial construct, and many current taxonomic classifications have a limited genetic basis. Thus, species diversity is itself an imperfect surrogate for genetic diversity. Third, biodiversity encompasses variability at all levels of ecological organization. Although species are the principal focus of biodiversity discussion and debate, species diversity remains only one component of biodiversity. Preserving habitat diversity would enable extant populations to evolve under variable environmental conditions. Finally, the most important lesson learned from our national experience with endangered species preservation is that habitats—not species—are the appropriate conservation management unit.

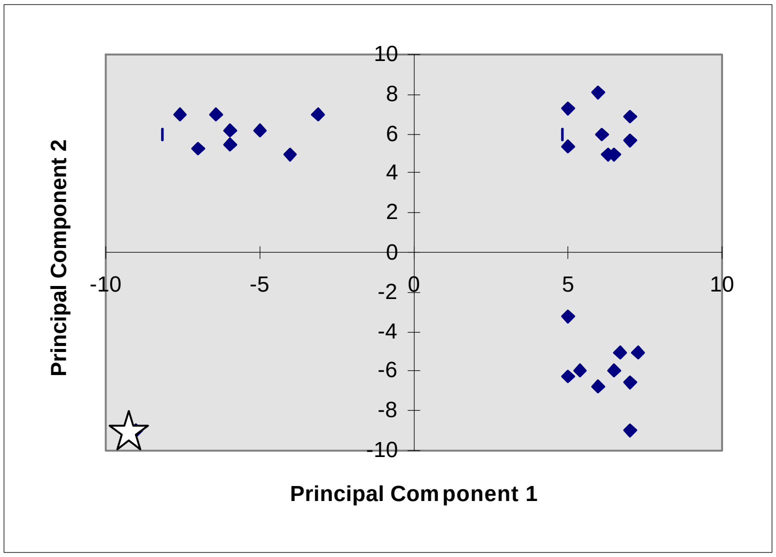
Regardless of whether USDA chooses to adopt habitat diversity as a suitable proxy for biodiversity, the question of operationalizing the habitat diversity assessment endpoint remains. To that end, note that the assemblage of sites protected under the CRP and other conservation programs constitutes a habitat conservation network. To achieve the objective of maximizing habitat diversity, we need to maximize the combined “dissimilarity” among sites included in the conservation network. Each of the sites may be described using readily available or estimable physiographical data (e.g., elevation, temperature, precipitation, soil pH, etc.) and ecosystem structure data (e.g., density and height of dominant vegetation, biomass per unit area, etc.). Statistical ordination procedures such as principal components analysis (PCA) can project the resultant n-dimensional data swarm into a more manageable two- or three-dimensional space. (Essentially, procedures such as PCA collapse multiple variables into a composite of variables related in a consistent fashion to the data. For example, temperature and precipitation could “map-down” to a single component that indicates available moisture. See Pielou (1984) for further details.) The distance between conservation sites mapped into this reduced coordinate framework represents a measure of dissimilarity.

To illustrate the ordination approach, assume that the sites currently enrolled in the habitat protection network are located in the northwest, northeast, and southeast quadrants of the PCA scatter plot in Figure 3. Adding the site in the southwest quadrant (indicated by the star) to the network would increase the diversity of habitats under protection more than adding an additional site in any of the other three quadrants. This very simple example

[10] Andrew Solow of Woods Hole Institute, Martin Weitzman of Harvard, Stephen Polasky of Oregon St., and others have applied statistical ordination techniques to measure species diversity in an attempt to develop methods for optimizing the biodiversity of protected area networks.

conceptually demonstrates the utility of the ordination approach. The solution would be much messier in practice, however, because habitat diversity would have to be maximized under multiple constraints that may be hard to define—e.g., limiting the habitat conservation network to sites of a minimum "viable" size and establishing an appropriate level of connectivity among habitat patches.[11] Furthermore, the ordination approach would have little practical utility if the first two or three principal components fail to account for a majority of the variability (as indicated by the available data) among sites.

Figure 3. Conceptual Diagram for Maximizing Habitat Diversity



7. Evaluate expert opinion about what constitutes an adverse effect. As noted above, environmental policymaking often requires a determination of what constitutes an adverse health or environmental effect. Just as frequently, however, it is difficult to specify a threshold or discontinuity along a continuum of observed physical or biological changes that can distinguish adverse from harmless or negligible responses to environmental stressors. For example, detectable infection from drinking water pathogens is associated with human health responses ranging from asymptomatic to severe gastrointestinal illness. Similarly, erosion varies in intensity, and identifying the threshold between sustainable and unsustainable erosion levels is fraught with uncertainty. In such cases, therefore, environmental policymakers decide, implicitly or explicitly, what level or type of systemic response to environmental stressors public policy should seek to avoid.

[11] To illustrate how hard it might be to define these constraints, consider that maintaining corridors between protected areas enables the movement and spread of non-indigenous species, pests and pathogens as well that of desired indigenous species.

To provide guidance to conservation program decisionmakers, one might survey experts regarding what they consider to be adverse effects on soil, plants, wildlife, water quality, etc. It is unclear, however, whether such a survey would a) elicit expert judgment under conditions of scientific uncertainty, b) simply obtain the value-based opinions of a specific sub-population, or c) (perhaps most likely) produce a confounded admixture of both. Soil, plant, and water scientists, ecologists, and conservation professionals are clearly more capable than the lay public of responding to survey questions seeking well-defined responses in terms of agricultural conservation program risk assessment endpoints. However, there are legitimate concerns that may be raised about such a survey. Experts are not uniquely qualified to determine what environmental risks public programs should seek to avoid, and some would argue that a distribution of “expert” opinion on what constitutes an adverse effect represents a skewed sample of the preferences of the general population. There are also concerns that many such experts are heavily invested professionally and intellectually in particular positions on the issue. Motivated by the policy, research funding, and other implications, some respondents may seek to bias the outcome or engage in strategic behavior intended to “offset” the perceived biases of other experts. Possible results of motivational biases would include an exaggeratedly long- or heavy-tailed distribution of expert opinion that would diminish the value of the survey information to conservation program decisionmakers. On the other hand, quantitatively or qualitatively different scientific models may underlie differences of expert opinion regarding the adversity of different levels of a risk assessment endpoint. In this case, a survey of experts’ subjective beliefs may be an appropriate and valuable decisionmaking input.

In the Appendix, we elaborate a statistical procedure (based on a robust confidence interval for location) that may be used to evaluate expert opinion about what constitutes an adverse effect. The procedure is intended to: 1) summarize survey responses for communication to decisionmakers when an expert consensus exists; 2) identify when there is a lack of expert consensus; and 3) safeguard against danger that some experts might intentionally exaggerate their opinions for strategic purposes.

8. Be specific about defining endpoints and measures. We noted that several of the assessment endpoints and evaluation measures lacked specificity, or were too numerous to be terribly valuable. Suggested here are some approaches to increasing specificity of endpoints.

- *Regarding soil quality*: Over the long term, the NRC Committee on Long-Range Soil and Water Conservation recommended replacing the soil loss tolerance (T) with a more robust set of soil quality indicators but found that in the short term, the soil loss tolerance level is the best standard available (NRC 1993). The Committee recommended that USDA (and EPA) develop a minimum data set of soil quality indicators, standardized methods for their measurement, and standardized methods to quantify changes in soil quality. The Committee’s “minimum data set” would include: nutrient availability, amount of organic carbon, amount of labile carbon, texture, water-holding capacity, structure, maximum rooting depth, salinity, and acidity/alkalinity. NRC (1993) discussed many composite soil quality indicators provided by the scientific literature, but the Committee failed to make a specific recommendation. Therefore, it is fair to say that USDA has considerable latitude in coming to closure on a set of soil quality indicators that monitor soil ecological functions other than plant growth promotion.

- *Evaluate the several measures of soil quality, to see if one or two provide most of the information needed.* For the purposes of national conservation program targeting and assessment, USDA should consider sacrificing depth for coverage in the minimum soil quality data set due to budget constraints. USDA might find, for example, that nutrient availability, rooting depth, soil porosity, and organic matter content are generally adequate for most agricultural regions. Salinity and acidity/alkalinity might be monitored only in regions where these soil properties are recognized problems. Given the increased emphasis on off-site damages associated with agricultural practices, priority should be given to selecting soil quality indicators that provide information about runoff and leaching potential that is independent of the information already provided by erosion indices.

- *Regarding pesticides:* It seems that much information is lost when pesticide loadings are simply aggregated in bulk across all pesticides. At a minimum USDA can readily weight pesticides loadings by use of toxic equivalents. For example, pesticide toxicity weights could be normalized by means of an indicator of acute aquatic lethality such as the LC_{50} (the pesticide concentration that is lethal to 50% of aquatic test organisms over a short duration).

- *Regarding low-probability, high consequence risks*: Generally speaking, environmental indicators based on average conditions can be misleading. Indicators should take into account the fact that relatively infrequent but severe events (e.g., intense storms, wildfires, severe droughts and floods) can be responsible for a disproportionate share of natural resource damages.

Appendix: A Procedure for Evaluating Expert Opinion

The proposed procedure for evaluating expert opinion about what constitutes an adverse effect is a decision analysis model employing a robust confidence interval for location based on a Descending M-estimator. The procedure would be applicable in cases where responses to an environmental stressor can be quantitatively characterized along a single, continuous dimension (e.g., soil loss per unit area from erosion or the concentration of drinking water pathogens in stool samples). The procedure would be limited to cases where there are a substantial number of potential “experts” to survey (e.g., 50 NRCS State conservationists), and the expert survey respondents may be interpreted as either a sample, probably non-representative, of the general population, or as a significant portion of the small population of qualified experts. Although the procedure might be useful in cases where there are pre-existing benchmarks or threshold values, we expect that it would be most valuable for decisionmaking where there is a lack of precedent or convention. (An example of the former would be soil erosion where we suspect that expert responses about an “adverse level” would tend to cluster around T or multiples thereof. An example of the latter might be determining what wildlife habitat recovery duration should be considered virtually irreversible.)

The model is intended to: 1) identify an “efficient” trapezoidal uncertainty distribution (i.e., as far from a uniform distribution as is justifiable) to summarize survey responses for communication to decisionmakers when an expert consensus exists; 2) identify when insufficient expert consensus may exist to justify combining the survey results into a single uncertainty distribution (i.e., when the Descending M-estimator algorithm fails to converge to a unique solution); and 3) safeguard against potential motivational biases to furnish extremely non-central survey responses.

It seems likely that the distribution of expert opinion regarding what constitutes an adverse environmental effect would violate assumptions of normality and symmetry. In general, the maximum likelihood estimate (MLE) can be made robust, or modified to reduce its sensitivity to outliers and yet still retain high efficiency at the ideal, assumed model. Huber (1964) modified the sample mean (the MLE of the population mean μ under the normal distribution) to minimize asymptotic variance, where the maximum is taken over all symmetric distributions which are close to the normal distribution. This estimator is now called Huber’s M-estimator (Ruppert, 1985). The M-estimator can be expressed as a weighted average and is solved as an iterative minimization problem.

M-estimators that assign no weight to all very large deviations have been recommended since these ignore gross outliers, rather than merely bounding their influence. Such estimators are called Descending M-estimators. If the normal distribution is contaminated by an asymmetric distribution, the problem of estimating central location becomes somewhat ambiguous. Huber (1981, pp. 74-76) shows that the sample median solves the problem of minimizing the maximum asymptotic bias. However, the minimax risk strategy can be unnecessarily conservative, and it can be worthwhile to increase the maximum risk slightly beyond its minimax value to gain better performance of the estimator (i.e., less variance) at longtailed distributions. This can be accomplished by using a descending M-estimator (Huber, 1981, pp. 100-103). Tukey's confidence interval for the sample median (based on Wilcoxon's signed rank statistic) also presumes a symmetric distribution (Hollander and Wolfe, 1973, pp. 33-50). Therefore, a descending M-estimator is well suited for characterizing the central location of distributions that are somewhat non-normal or asymmetric.

The robust confidence interval procedure developed by DuMond and Lenth (1987) is based on a Descending M-estimator (the biweight), has high efficiency in the normal case and maintains high validity over a broad range of distributions (uniform, symmetric heavy-tailed, and skewed heavy-tailed).[12] The interval is expressed as:

$$M \pm t_{1-\alpha/2;\, n-1}[s/(\Sigma_n W_i)^{½}]$$

where $t_{1-\alpha/2;\, n-1}$ is the $(1-\alpha/2)$th quantile of the t distribution with n-1 df and s is the estimated standard error.

The biweight function (also called the bisquare) is given by:

$$\begin{aligned} w(t) &= \{1 - (t/k)^2\}^2 \text{ if } |t| < k \\ &= 0 \qquad\qquad\quad \text{ if } |t| \geq k \end{aligned}$$

where t is a standardized deviation from central location. Using a tuning constant (k) of 4.69 makes the efficiency of the biweight M-estimate relative to the sample mean (as a ratio of asymptotic variances) equal to 0.95 when the population is normal (DuMond and Lenth, 1987). Table 1 compares some weights assigned by the biweight M-estimator (k=4.69) with Huber's M-estimator (k=1.5) and a trimmed mean (with $w(t) = 0$ for $|t| \geq 3$). Although the Biweight function assigns zero weight to some extremely non-central (in this case, $t \geq 4.69$) responses to offset the risk of motivational bias, because the weights assigned by the Biweight function decline smoothly and continuously, it can be regarded in some respects as a "fairer" rule for downweighting non-central observations (i.e., survey responses) than a discontinuous function (Huber's M-estimator or the Trimmed mean).

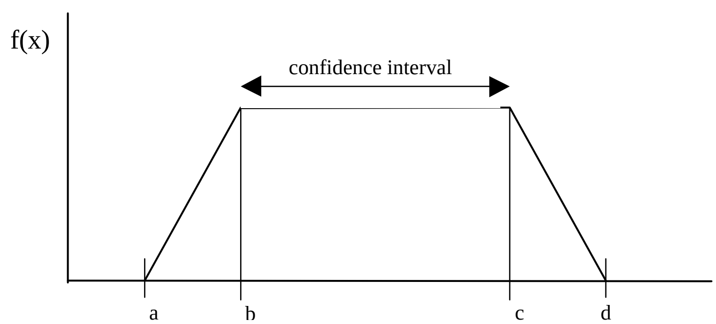
Table 1. **Comparison of weights (w(t)) assigned by the Biweight, Huber's and Trimmed Mean.**

	w(t)		
t	Biweight	Huber's	Trimmed
0.00	1.00	1.00	1.00
1.00	0.91	1.00	1.00
1.50	0.81	1.00	1.00
2.0	0.67	0.75	1.00
3.00	0.35	0.50	0.00
5.00	0.00	0.30	0.00

12 The procedure becomes somewhat unsafe when the population is both skewed and light-tailed.

The robust model for evaluating expert opinion would identify a trapezoidal uncertainty distribution to summarize survey responses regarding what constitutes an adverse health or environmental effect for communication to decisionmakers. Attractive features of the trapezoidal distribution (see Figure 4 below) are that it is more informative than a uniform distribution, exhibits a rough similarity to a Gaussian distribution, and (unlike the triangular distribution) does not require the specification of a singular modal value. However, specifying the endpoints (a,d) and turning points (b,c) of the distribution with precision often requires more information than may be available to the analyst (Seiler and Alvarez 1996). To identify the range of equally most likely values in the trapezoidal uncertainty distribution, the model will employ the confidence interval of location described above. The efficiency and robustness of the M-estimator procedure ensures that the trapezoidal distribution will be as informative (i.e., as far from a uniform distribution) as is appropriate while maintaining good validity over a broad range of underlying distributions of expert opinion. The endpoints of the distribution would be the most extreme (i.e., non-central) non-zero weighted responses.

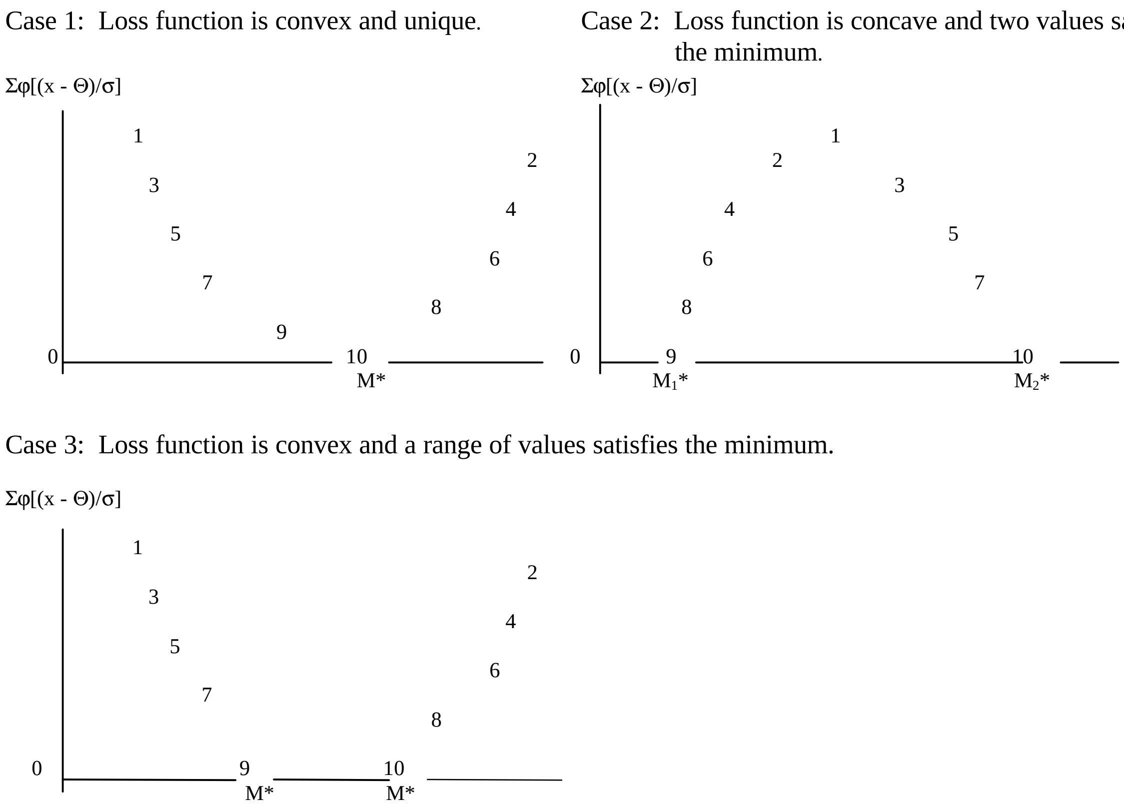
Figure 4. Probability density function for a trapezoidal distribution describing the distribution of expert opinion on the percent lung function decrement regarded as adverse.



Valid objections can be raised about assigning variable "expert weights" to pool expert responses into a combined distribution. Genest and Zidek (1986) demonstrate that imposed consensus functions require "dictatorial" aggregation methods ($w_i = 1$ for some *i* and 0 for others) and note their "impossibility" due to their unsatisfactory nature. However, the proposed model for evaluating expert opinion does not impose consensus. It provides a means of identifying whether expert consensus occurs, and if so, describing that consensus in a manner that is both informative (i.e., efficient) and robust (i.e., to deviations from the normal model and to motivational biases).

With a Descending M-estimator, there will be multiple solutions to the minimization problem when there is insufficient central mass in the weighted probability distribution. Recall that computation of an M-estimate is an iterative minimization problem. If M's loss function is convex (successive iterations converge) with respect to the summation on the vertical axis of Figure 5, the minimizer may or may not be unique. If M's loss function is concave with respect to the summation, the minimizer is definitely non-unique. (The numbers in the graphs in Figure 5 refer to the iterative sequence.)

Figure 5. M-estimate uniqueness



Case 1: Loss function is convex and unique.

Case 2: Loss function is concave and two values satisfy the minimum.

Case 3: Loss function is convex and a range of values satisfies the minimum.

While this ambiguous property of Descending M-estimators may be a disadvantage in some contexts, it is a strength in the context of evaluating expert opinion because it means that the procedure does not impose a consensus where one may not exist. Conversely, when the procedure does yield a unique solution, there is increased confidence that an expert consensus can be identified and described in a manner that will not overload the decisionmaker with information. An explicit determination of a lack of consensus among experts can be an important finding by itself. If there is a lack of consensus among the experts regarding what constitutes an adverse effect, one possible explanation is that experts are using quantitatively or qualitatively different biological models in forming their responses. Alternatively, it may signify polarized value-based opinions in the expert subpopulation. It should be noted that a determination of expert consensus could also reflect shared value-based opinions in the subpopulation.

In terms of addressing potential sources of bias among experts, existing uncertainty and decision analysis methods focus primarily on preventing or correcting for expert overconfidence (see Morgan and Henrion, 1990). To counteract potential motivational biases, analysts may rely on procedural means to assure "balance" among experts. However, such efforts to avoid biasing the central tendency of expert opinion can easily bias the spread of the distribution. This proposed methodological approach could be used in conjunction with procedural means for addressing motivational bias in eliciting expert judgment to provide a robust *and* efficient description of the distribution of expert opinion.

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