Web Appendix A
Simulating Working-Land Payment Programs

A.1 Regional Simulation Model

To evaluate the economic and environmental implications of alternative WLPPs, we employ a regional, agricultural-sector model for the United States. This is a comparative-static, spatial and market-equilibrium model, which incorporates agricultural commodity, supply, demand, environmental impacts, and policy measures (House et al., 1999). The model includes 45 geographic sub-regions, 23 production inputs, and the production and consumption of 44 agricultural commodities and processed products.[1] More than 5,000 crop production enterprises at the sub-region level are differentiated according to cropping rotations, tillage practices, and fertilizer rates; 90 livestock and poultry production enterprises are delineated at the region level by species. Production levels and enterprises are calibrated to regularly updated production practices surveys using a positive math programming approach (Howitt, 1990), the USDA multiyear baseline (USDA, 2001), and the National Resources Inventory (USDA, 1994).[2]

Changes in production are in turn linked to the potential environmental changes via the Environmental Policy Integrated Climate (EPIC) Model. The model simulates daily weather, hydrology, soil temperature, erosion-sedimentation, nutrient cycling, tillage, crop management and growth, and pesticide transport to the edge of the field (Mitchell et al., 1998). Crop yields and environmental externalities are estimated on a per-acre basis for short-run (mean over 7 years) and long-run production (mean over 67 years) given historical climate and soils data from across the United States. The yield data are combined and calibrated to current production patterns. For certain pollutants (e.g., nitrogen, phosphorus, soil sediment, and pesticides) a runoff transport component is calibrated to observed pollutant levels using estimates from the U.S. Geologic Survey (Smith et al., 1997) in order to estimate instream environmental effects of agricultural production.

A.1.1 Baseline Production

The simulation model is first calibrated to projected production patterns (USDA, WAOB, 2003), solving for optimal regional (subscript *k*) production levels for cropping enterprises (X_{ki}) and livestock activities (X_{kl}):

$$\text{(eq A.1)} \quad \max_{X_{ki},X_{kl}} \sum_{ki} (P_i - VC_{ki}) X_{ki} + \sum_{kl} (P_l - VC_{kl}) X_{kl} \, .$$

Here P_i is the equilibrium price vector for cropping system *i*, and P_l are equilibrium prices for livestock; VC_{ki} and VC_{li} represent regional variable costs of production.

---

[1] The model accounts for production of the major crop (corn, soybeans, sorghum, oats, barley, wheat, cotton, rice, hay, silage) and livestock (beef, dairy, swine, and poultry) categories comprising approximately 75 percent of agronomic production and more than 90 percent of livestock production in the United States (USDA 1997). We do not consider potential applications of manure to rangeland, vegetable, horticulture, sugar, peanut, or silviculture operations.

[2] This model has been used to examine other agri-environmental policies (Johansson and Kaplan, 2004; Claassen et al., 2001), climate change mitigation (Lewandrowski et al., 2004), water quality policy (Ribaudo et al., 2001), wetlands policy (Heimlich et al., 1997), and sustainable agriculture policy (Faeth, 1995).

The acreage constraints imposed under the policy simulations are represented by:

(eq A.2) $\sum_{i} X_{ki}^{0} = \sum_{i} X_{ki} \ \forall k$,

where $\sum_{i} X_{ki}^{0}$ is the amount of cropland acres in region *k* before implementing the WLPP and $\sum_{i} X_{ki}$ is the amount of cropland acres in region *k* after implementing the WLPP. In other words, producers cannot receive program payments for environmental benefits generated from retiring land from production or for land that had not previously been cropped prior to the WLPP implementation.

A.1.2 Practice-Based Agri-Environmental Payments

Those management practices that are targeted towards reducing soil erosion and generally improving water quality are modeled (table A.1).[3] These practices represent approximately 90 percent of the non-livestock, non-structural EQIP contracts between 1997 and 2000. Practice costs are calculated assuming a 3-year implementation period.[4] The 3-year total cost is then discounted at 7 percent over a 10-year contract period. In addition to management practices, "base payments" are included in the program payment, structured to resemble the tiered system of payments found in the Conservation Security Program. Base payments are pegged to regional crop rental rates and are calculated to represent a 10-year net present value of average rental rates for cropland (Farm Service Agency, 2003). [5]

Table A.1. Practice-based conservation payments (per acre)

Eligible practices	Farm production region[a]									
	AP	CB	DL	LA	MN	NT	NP	PA	SE	SP
Base payment[b]	2.08	3.84	1.95	2.74	0.84	1.55	1.57	2.63	1.30	0.87
Conservation rotation[c]	2.81	1.41	1.40	1.40	1.83	1.87	1.40	1.88	2.81	1.87
Nutrient management[d]	2.81	2.25	1.40	1.68	2.81	2.25	1.24	2.81	2.81	4.49
Hay[e]	30.32	21.15	28.38	19.91	10.95	15.36	10.71	17.41	30.89	13.48
Mulch till[f]	2.81	2.25	2.81	3.37	2.81	3.37	1.63	1.68	8.42	2.62
No-till[g]	2.81	2.25	4.21	2.81	4.21	3.37	3.37	5.62	5.62	2.62

a/ Appalachia (AP) = KY, NC, TN, VA, WV; Corn Belt (CB) = IA, IL, IN, MO, OH; Delta States (DL) = AR, LA, MS; Lake States (LA) = MI, MN, WI; Mountain (MN) = AZ, CO, ID, MT, NM, NV, UT, WY; Northeast (NT) = CT, DE, MA, MD, ME, NH, NJ, NY, PN, RI, VT; Northern Plains (NP) = KS, ND, NE, SD; Pacific (PA) = CA, OR, WA; Southeast (SE) = AL, FL, GA, SC; Southern Plains (SP) = OK, TX.
b/ Base pay values derive from mean rental rates for non-irrigated cropland under the Conservation Reserve Program (FSA, 2003) multiplied by 5 percent to correspond to a tiered structure described in Chapter 4.
c, d, e, f, g/ The reported payments are the median contract values for EQIP calculated to reflect a 100 percent cost share (Cattaneo, 2003).

[3] Note that the cost of these practices, the benefits provided, and the associated rental rates are often not correlated such that practice-based conservation payments solicit the most cost-effective environmental benefits (see Web Appendix C).

[4] This follows the benefit-cost methodology used by USDA (NRCS, 2003).

[5] Base payments in the Conservation Security Program increase with tiers. At the lowest tier, producers receive cost-share plus a base payment of 5 percent of the land rental rate. This rate increases to 15 percent at the highest tier of participation.

More formally, the working-land program payment (P_{ki}) for an eligible practice (k) in region (i) will be a function of the cost of implementing the eligible practice in that region (EP_{ki}), the percentage of that cost that is reimbursed under the program, and the base payment amount determined from regional crop rental rates. Cost-share percentages are chosen to be at the 50-percent level as found in the 2002 EQIP guidelines (USDA, NRCS, 2003). The program payment for any given eligible practice in our simulation model can be written:

(eq A.3) $P_{ki} = (0.5 \times EP_{ki}) + (Base\_Pay_i)$.

The total agri-environmental payment (AEP) available for a producer in region (i) for eligible practices (k) is then:

(eq A.4) $AEP_{ki} = \sum_k P_{ki}$.[6]

A producer can receive higher payments by combining several cropping production practices. For example, in our simulation, a producer in the Corn Belt could receive practice-based payments of \$13.53 per acre for a no-till (\$2.25 per acre) or mulch tillage cropping system that included nutrient management practices (\$2.25 per acre) and hay rotation (\$21.15), which is also a conservation rotation (\$1.41 per acre) at a 50-percent cost-share rate. In addition, the producer would be eligible under this program for a base payment of \$11.52 per acre (or 3 × \$3.84) for a total of \$25.05 per acre.[7]

A.2 Policy Simulations

A.2.1 Good-Act

Under this scenario, farmers already employing eligible practices (good actors) are eligible to receive WLPP payments along with an additional payment based on a regional land rental rate. Various levels of an exogenously determined budget (B) are simulated, restricting total payments so that the budget is not exceeded. Regional program payments are further restricted to be a percentage of the total budget ($distrib_k$), which is equal to the distribution percentage of regional EQIP payments:

(eq A.5) $B \times distrib_k \geq Perc_k \sum_i X_{ki} AEP_{ki} \ \forall k$,

where X_{ki} is the acreage level of the eligible practices after the WLPP is offered, and *Perc* is the optimal percentage of acres that actually receive agri-environmental payments (AEP_{ki}) to meet the regional budget constraint. This distributional constraint is imposed to insure that program payments are spread across the entirety of U.S. cropland. The resulting optimization is:

(eq A.6) $\max_{X_{ki}, X_{kl}} \sum_i (P_i - VC_{ki} + AEP_{ki}) X_{ki} + \sum_l (P_l - VC_{kl}) X_{kl}$

subject to eq A.2 and eq A.5.

---

[6] Note that producers can receive two payments for incorporating hay into their rotation (*conservation rotation* and *hay*), but can only receive 1 × *base payment* for this combination.

[7] This is an upper bound on per acre payments for this combination of practices as the payment of \$21.15 for planting hay will be multiplied by the share of hay in a particular rotation. For example, a continuous hay rotation would receive the full \$21.15 payment (at a 50 percent cost-share rate), whereas a corn-soybean-hay rotation would receive \$6.98, or 0.33 × \$21.15 (also at a 50 percent cost-share rate).

A.2.2 Practice

Next, restrict eligibility for the practice-based payments (and base payments) to those farmers that adopt new practices:

(eq A.7)

$$\max_{X_{ki},X_{kl}} \sum_{ki}(P_i - VC_{ki})X_{ki} + \sum_{ki}(X_{ki} - X_{ki}^0)AEP_{ki} + \sum_{kl}(P_l - VC_{kl})X_{kl},$$

subject to eq A.2 and to

(eq A.8) $B \times distrib_k \geq Perc_k \sum_i (X_{ki} - X_{ki}^0)AEP_{ki} \ \forall k$.

A.2.3 Performance

Under this scenario, payments are simulated for reducing the number of Aggregate Environmental Index points, AEI_{ki} (see Web Appendix B), generated from crop production. No good-actor provisions or base payments are attached to these contracts. Furthermore, the distributional budget constraint from above is relaxed, as producers are able to garnish payments for environmental points broadly defined to include nine environmental criteria. The optimization model for this scenario is depicted by:

(eq A.9)

$$\max_{X_{ki},X_{kl}} \sum_{ki}(P_i - VC_{ki})X_{ki} + PPT\sum_{ki}(X_{ki}^0 - X_{ki})AEI_{ki} + \sum_{kl}(P_l - VC_{kl})X_{kl},$$

subject to

eq A.10) $B \geq PPT\sum_{ki}(X_{ki}^0 - X_{ki})AEI_{ki}$ and to eq A.2,

where *PPT* is the agri-environmental price per point offered under the program. A national price for environmental performance points is assumed, which could just as easily be specified on a regional basis.

A.2.4 Bid

To capture the fact that it is cheaper to achieve some benefits points than others, which would be reflected by bidding provisions, the area under the payment-marginal benefits curve distilled from the performance-based policy is integrated to determine aggregate program payments. Essentially, as the national price for environmental performance increases, an increasing number of farmers will be willing to accept performance-based contracts. Here WTA_{ki} replaces *PPT* as the per-point payment level each enterprise would accept to generate environmental benefits:

(eq A.11)

$$\max_{X_{ki},WTA_{ki},X_{kl}} \sum_{ki}(P_i - VC_{ki})X_{ki} + WTA_{ki}\sum_{ki}(X_{ki}^0 - X_{ki})AEI_{ki} + \sum_{kl}(P_l - VC_{kl})X_{kl}$$

subject to (eq A.12) $B \geq WTA_{ki}\sum_{ki}(X_{ki}^0 - X_{ki})AEI_{ki}$ and eq A.2.

A.2.5 Hurdle

To model hurdle rates, payments are simulated for reducing the number of Aggregate Environmental Index points (see Web Appendix B) generated from crop production above and beyond a pre-determined reference level. The optimization model for this scenario is depicted:

(eq A.13)

$$\max_{X_{ki},X_{kl}} \sum_{ki}(P_i - VC_{ki})X_{ki} + PPT\sum_{ki}[\max(0,(\overline{AEI} - AEI_{ki})]X_{ki} + \sum_{kl}(P_l - VC_{kl})X_{kl}$$

,

subject to eq A.2 and to

(eq A.14) $B \geq PPT\sum_{ki}[\max(0,(\overline{AEI} - AEI_{ki})]X_{ki}$,

where PPT is the agri-environmental price per point offered under the program for practices above a pre-determined reference level, $\overline{AEI}$ (recall that the lower the AEI_{ki} score, the better its environmental performance).

Web Appendix B
Aggregate Environmental Indices

B.1 Scoring Production Systems

An aggregate benefits score (AEI_{ki}) is generated for each cropland acre (*i*) and region (*k*). This aggregate benefits score is composed of the "relative damage estimate" (RDE_{kji}) for each of the environmental externalities (*j*) based on the mass of each pollutant that potentially arrives at the appropriate medium from cropping system (*i*) and region (*k*). The respective RDEs are the product of edge-of-field emissions and the corresponding transport factors:

(eq B.1) $RDE_{kji} = q_{kji} * t_{kj}$,

where *q* represents edge-of-field emissions and *t* represents the relevant transport factor. Transport factors are calculated using USGS-estimated agricultural discharge in the case of surface water pollutants,[8] and are assumed to be 100 percent for air emissions and leaching (i.e., there is no assumed loss in mass from the edge-of-field emissions to the relevant destination media).

Summing over current production levels provides an estimate of the potential discharge of these externalities, which vary considerably by region (table B.1). For example, the largest amount of sediment and nutrients are discharged from the Corn Belt, which has the most production acres of all regions. However, pesticide leaching to groundwater is highest in the Lake States region, where the underlying topography makes it relatively more susceptible to leaching. Nitrogen leaching is highest in Appalachia.

Table B.1. Baseline values for environmental indicators by region[a]

	Air		Soil	Groundwater		Surface water			
	Carbon[b]	Wind	Productivity[c]	Pesticides[d]	Nitrogen	Pesticides	Sediment	Nitrogen	Phosphorus
Region	(metric tons)	(Tons)	($)	(TPUs)	(Lbs.)	(TPUs)	(Tons)	(Lbs.)	(Lbs.)
NT	3	1	15	3,859	130	8,539	8	173	12
LA	12	113	11	11,942	357	27,217	20	358	15
CB	39	42	104	3,706	234	102,671	102	1,484	105
NP	24	120	102	1,272	112	21,583	15	407	24
AP	4	1	42	8,862	400	24,025	12	284	20
SE	2	0	1	4,526	182	17,847	12	116	12
DL	9	0	55	825	141	61,899	10	236	14
SP	13	185	3	916	63	103,250	17	234	15
MN	6	227	8	399	31	108,813	12	119	6
PA	2	29	30	13	55	54,173	5	89	2
US	114	718	372	36,322	1,706	530,017	213	3,499	225

a/ Environmental indicators are measured in millions of units discharged from cropland, not inclusive of animal production.
b/ Carbon emissions are calculated according to the Intergovernmental Panel on Climate Change estimates (IPCC, 1996). The values indicate the amount of carbon emitted when converting land from native pasture.
c/ Loss in soil productivity is measured in lost net present value of crop output per acre over a 60-year time horizon due to soil degradation.
d/ TPUs refer to “toxicity persistence units” (Barnard et al., 1997). These refer to the sum of reference doses (maximum daily human exposure resulting in no appreciable risk) of the pesticides used for a particular cropping enterprise multiplied by the number of days, each of those pesticides remains active in the environment. As a point of reference the number of TPUs in a pound of DDT = 4,443 million and in a pound of Borax = 103,872.

---

[8] Estimates of phosphorus and nitrogen discharge are found in Smith et al. (1997). Transport of nitrogen to estuaries is found in Alexander et al. (2000). Transport factors for surface water pesticides and sediment are assumed to be similar to phosphorus transport.

Production systems with low relative damage estimates (RDEs) indicate cleaner practices; those with high RDEs contribute higher quantities of pollutants to the environment. To characterize each crop production system (i) and its potential to generate environmental benefits in each region (k), the relative damage estimates (RDE_{kji}) are converted to a 0-1 impact index (I_{kji}) for each pollutant (j):

(eq B.2) $$I_{kji} = \left(\frac{RDE_{kji} - \min(RDE_j)}{\max(RDE_j) - \min(RDE_j)} \right),$$

where $\min(RDE_j)$ and $\max(RDE_j)$ are the minimum and maximum damage estimates across all systems (i) and regions (k) for the jth environmental pollutant. For example, the potential to deliver nitrogen to groundwater is highest for conventionally tilled, soybean-wheat rotations on non-highly erodible land in the Lake States production region (65.83 lbs./acre/year). Its benefit index value for nitrogen loading to groundwater is therefore normalized to 1.0.

Normalizing potential discharge in this manner implies a point equivalency ratio between the nine pollutants:

(eq B.3)
$$\max(RDE_m) - \min(RDE_m) : \max(RDE_n) - \min(RDE_n) \ \forall\, m \neq n \in j \, .$$

The point equivalency values reflect equivalent amounts of each pollutant necessary to generate 1 unit of I_{ki}. For example, the point equivalency ratio between nitrogen and phosphorus discharge is approximately 10.93, which implies that the maximum potential reduction in nitrogen discharge given the range of practices in the simulation model divided by the maximum potential reduction in phosphorus discharge is 10.93.

The individual indicators are combined to generate an aggregate environmental index score (AEI_{ki}) specific to each production system and region that reflects the total management effects of that production system on the environment:

(eq B.4) $$AEI_{ki} = f(I_{kji}) \, .$$

Several functional forms have been promoted to construct aggregate measures of environmental quality from individual indices (Heimlich, 1994). This report uses a weighted sum of the individual environmental indicators as an aggregate environmental quality index:

(eq B.5) $$AEI_{ki} = \sum_j w_{kj} I_{kji} \, ,$$

where w_{kj} are weights on pollutant damages. This functional form implies that damages to the environment are continuous and linear in discharge. This is similar to other aggregate measures of environmental quality such as the Environmental Benefits Index, or EBI (USDA, Farm Service Agency, 2002) and the Index of Watershed Indicators (U.S. EPA, 2002).[9] Ideally, the weights chosen would reflect socioeconomic preferences for mitigating the various pollutants. We develop several weighting schemes to illustrate how such preferences may result in different program outcomes.

B.2 Weights

Developing weights that reflect society's preference for different environmental benefits is difficult. One means to weight multiple criteria is to assign monetary values to changes in the amount of pollutants released into the environment -- increased levels of reduction are associated with higher environmental benefits and associated monetary value (see, for example, Hansen et al., 2002). Many studies have asked survey respondents how much they are willing to pay for a reduction in their exposure to certain chemicals. Examples include nitrates in drinking water supplies (Crutchfield et al., 1997); fertilizers, pesticides, and manure in surface water resources (Hite et al., 2002; Stumborg et al., 2001; Van Kooten et al., 1998). Others have used travel cost methods to determine how valuable variable recreation opportunities are to the public (e.g., sediment loads and fishing recreation in Feather et al., 1999) or hedonic analysis to reveal how preferences of consumers are affected by variable environmental quality (e.g., sulfur and nitrogen in the air and its affects on housing prices in Kim et al., 2003). Because these studies are often site specific, many researchers impute the estimated values to other regions or populations using a method termed "benefit transfer" (see also Web Appendix C). This saves on the cost of designing and implementing new surveys, but is less accurate than an original survey. Examples include nutrient loads in the Chesapeake Bay (Morgan and Owens, 2001), nitrate in drinking water (Crutchfield et al., 1997), and sediment loads in U.S. surface waters (Feather and Hellerstein, 1995).[10]

These studies raise several important questions. First, it is clear that there are many benefits to consider when examining the value of reducing pollutant levels, including human health benefits (e.g., reduced exposure to toxic chemicals), recreational benefits (e.g., the oft cited "swimmable, boatable, fishable" standard found in the Clean Water Act), and ecological benefits (e.g., reduced probability of fish kills). Second, these studies often examine the value of improving a particular metric by a percentage, making it difficult to decipher the value per physical unit of pollutant, suggesting that per-unit benefits will depend on the level from which the change is occurring.

In the absence of a national or local survey that explicitly asks such questions, this report adopts an approach using data about how program decisionmakers have valued past efforts at addressing multiple environmental criteria. How public preferences translate into program expenditures and mandates is well documented (Variyam and Jordan, 2001; Besley and Burgess, 2002; Dixit et al., 1997; Crémer and Palfrey, 2002). Looking explicitly at conservation programs Bastos and Lichtenberg (2001) noted that incentive payments are linked to public preferences for environmental quality. Moreover, while the link between policy expenditures for working-land payment programs, environmental standards, and public preferences may not be completely transparent,

---

[9] The assumptions of continuous and linear damages serve to illustrate the costs to producers in reducing the physical amounts of these pollutants from entering the environment. More complicated damage functions can be incorporated by changing the form of the aggregate environmental indicator.

[10] A summary of these methods can be found online at http://www.ecosystemvaluation.org/ (King and Mazzotta, 2003).

Reichelderfer and Boggess (1998) noted that program decisionmakers can learn and improve the cost-effectiveness of conservation program controls.

B.2.1 National-level weights

Environmental Benefits Index weights – The Conservation Reserve Program (CRP) was initially designed to reduce the quantity of soil erosion from cropland cultivation by encouraging U.S. farmers to "retire" lands with a high potential for soil erosion. Today, CRP contracts are evaluated at the national level using multiple environmental criteria found in the Environmental Benefits Index (EBI). The EBI in sign-up 26 (FSA, 2003), developed to score CRP contracts, includes weights for groundwater leaching, soil productivity loss, surface water discharge, wind erosion, and carbon sequestration. There are a total of 415 points available for any particular contract enrolled in the CRP (table B.2). However, this report does not consider wildlife benefits, water quality location, enduring benefits, or air quality zones in our analysis, which leaves a total of 230 points available for generating weights.[11]

Table B.2. Weights for an Aggregate Environmental Index (AEI) using the EBI

EBI Category	EBI	EBI	Weight
Wildlife			
N1a. Cover	50		
N1b. Enhancement	20		
N1c. Priority Area	30		
Total	100		
Water Quality			
N2a. Location	30		
N2b. Groundwater	25	25	0.11
N2c. Surface Water	45	45	0.20
Total	100		
Erosion (Soil Productivity)	100	100	0.43
Enduring Benefits	50		
Air Quality			
N5a&b. Wind Erosion	30	30	0.13
N5c. Air Quality Zones	5		
N5d. Carbon Sequestration	30	30	0.13
Total	415	230	1.00

Source: FSA(2004)

By mapping these weights to air, soil, groundwater, and surface water, a set of implicit weights is developed to broadly reflect the nine environmental criteria considered in this report. The EBI places a relatively large weight on maintaining long-term soil productivity (reducing soil erosion) and improving water quality (ground and surface), but values reductions in wind erosion and

---

[11] Because we do not have specific data about the effects of different cropping systems on these categories, it is unclear how they could map into the nine selected pollutants for this analysis. For example, points attributable to being located in a "water quality region" could map to either ground-water or surface water quality. The conservative mapping in table B-2 eliminates categories that do not directly correspond to the nine pollutants included in this report.

carbon sequestration to a lesser degree. However, these implicit weights leave the question of weights within media; e.g., nitrogen leaching versus pesticide leaching. For these within-group comparisons, data from EQIP contracts and from the U.S. Environmental Protection Agency (EPA) is used to distinguish constituents of groundwater quality (nitrogen and pesticide leaching) and of surface water quality (table B.3).

Table B.3. National-level weights from the EBI

Medium	Initial weights (%)	Resource concern	Weights (%)
Air	0.13	Carbon	0.13
		Wind	0.13
Soil	0.43	Productivity	0.43
Groundwater	0.11	Pesticides[a]	0.03
		Nitrogen[b]	0.07
Surface Water	0.20	Pesticides	0.01
		Sediment	0.10
		Nitrogen[c]	0.04
		Phosphorus[d]	0.04
	1.00	Total	1.00

a/ b/ The water quality weights are allocated to nitrogen and pesticide leaching calculated from EQIP data in table B.4.
c/ d/ The weights for nitrogen discharge and phosphorus discharge are calculated from EPA data in tables B.5, B.6, and B.7.

EQIP Payment Amount Weights – EQIP contracts account for a variety of "resource concerns," among them surface water (including pesticides, erosion, and nutrient discharge), groundwater quality (including pesticides and nutrient loading), air quality (including wind erosion), and soil quality (including maintenance of soil productivity). By examining EQIP contract amounts and the stated primary resource concern, different sets of weights can be derived for potential pollutants stemming from regional agricultural production. Therefore, a comparison can be made of the relative weights associated with how much a given management practice might be expected to address one externality or another (table B-4). These weights do not compare unit measures of pollutant abatement; e.g., 1 pound of nitrogen abated compared with 1 pound of sheet and rill erosion. Rather, they reflect the relative importance of the criteria based on the relative amounts of payments paid out to practices that addressed them. For example, because carbon sequestration is not a resource concern under EQIP, it receives a weight of 0.[12]

[12] Such weights may (partially) reflect the policymakers' preconception of the performance of the relevant conservation program. For example, EQIP does not specifically address carbon sequestration, but CRP does. That may not indicate that EQIP policymakers do not value carbon sequestration, but that they realize the EQIP does not have a comparative advantage in providing incentives for carbon sequestration; whereas land retirement under CRP might.

Table B-4. National-level weights using EQIP expenditures

Medium	Initial weights (%)	Resource concern	Weights (%)
Air	0.05	Carbon	0.00
		Wind	0.05
Soil	0.46	Productivity	0.46
Groundwater	0.16	Pesticides	0.05
		Nitrogen	0.11
Surface Water	0.33	Pesticides	0.02
		Sediment	0.17
		Nitrogen[a]	0.07
		Phosphorus[b]	0.06
	1.00	Total	1.00

Source: FSA (2002).
a/ b/ The weights for nitrogen discharge and phosphorus discharge are calculated from EPA data below (Tables B.5, B.6, and B.7).

Returning to nutrients (nitrogen and phosphorus), there is no distinction between the type of nutrient being addressed under the surface or groundwater quality resource concern in the EQIP database. It can be assumed that, for groundwater quality, the nutrient of concern is nitrogen, which can directly affect human health through impaired well-water quality. However, the discharge of nitrogen and phosphorus into surface waters both result in significant water impairments. Therefore, we look to EPA's published nutrient criteria for rivers, streams, lakes, and reservoirs (table B.5) to develop these weights. The nutrient criteria represent EPA recommendations to States and Tribes for use in establishing water quality standards consistent with section 303c of the Clean Water Act (CWA). Using these criteria we can also generate a recommended nitrogen to phosphorus ratio by region, an indicator often used to determine the eutrophic potential of a water body (e.g., Scasso et al., 2001).

Table B.5. EPA nutrient criteria

	Nitrogen criteria		Phosphorus Criteria	
	Lake	River	Lake	River
Eco-region	micrograms per liter		micrograms per liter	
1	660	310	55	47
2	100	120	9	10
3	400	380	17	22
4	440	560	20	23
5	560	880	33	67
6	780	2,180	38	76
7	660	540	15	33
8	240	380	8	10
9	360	690	20	37
10	570	760	60	128
11	460	310	8	10
12	520	900	10	40
13	1,270	1,140	18	15
14	320	710	8	31

Source: EPA (2003).

The criteria are mapped into the 10 Farm Production Regions used in this report using an area-weighted average. In addition, based on the percentage of lakes and rivers impaired in each region, the recommended nitrogen and phosphorus concentrations for lakes and rivers are weighted to derive a single value for nitrogen and phosphorus for each region. In all regions, there is an abundance of both nitrogen and phosphorus discharge. To attain the recommended criteria, the amount of nitrogen and phosphorus in the water would have to be reduced by more than 80 percent in all regions (table B.6). The ratio of nitrogen to phosphorus reduction gives an indication of the relative importance of reducing the two nutrients.

Table B.6. Regional nitrogen and phosphorus concentrations

	Nitrogen		Phosphorus	
	Recommended[a]	Estimated[b]	Recommended	Estimated
Region	micrograms per liter		micrograms per liter	
AP	458	2,766	18	286
CB	1,048	7,639	44	688
DL	583	8,940	63	488
LA	817	8,924	31	705
MN	417	44,301	22	6,131
NP	794	17,855	43	1,498
NT	393	2,950	14	302
PA	292	22,378	20	2,565
SP	577	12,597	36	1,514
ST	32	54	1	5
US	772	13,437	37	1,394

a/ Recommended represents a weighted average of river and lake criteria based on the percentage of assessed rivers and lakes in these regions that are listed as threatened or impaired under the Clean Water Act's 303d reporting protocol (EPA, 2003b)

b/ Current annual loadings are estimated by the USGS for nitrogen and phosphorus (Smith et al., 1997). Current water flow per region is estimated from EPA (1996) data.

For example, to reach recommended levels of nitrogen and phosphorus concentrations in Appalachia, reductions in nitrogen discharge should occur at a rate approximately 9 times that for phosphorus discharge (table B.7). Recall that the point equivalency ratio between nitrogen and phosphorus is 10.93 for the range of practices included in this report. That is to say, that by using points to measure the environmental performance of various practices, a ratio of N:P weights of 10.93 is implicitly assumed. Therefore, the recommended N:P ratios in table B.7 are normalized by 10.93 to develop regional and national weights for nitrogen and phosphorus abatement. These weights are then multiplied by the weight given to the resource concern "nutrients discharged to surface water" to generate the relative national-level weights for reductions of nitrogen and phosphorus runoff (tables B.3 and B.4).

Table B.7. Recommended reductions for N and P runoff

	Recommended reductions			Weights	
	N	P	N:P ratio	N	P
Region	micrograms per liter			%	
AP	2,308	268	8.61	0.56	0.44
CB	6,591	644	10.24	0.52	0.48
DL	8,357	425	19.67	0.36	0.64
LA	8,107	674	12.02	0.48	0.52
MN	43,884	6,109	7.18	0.60	0.40
NP	17,061	1,455	11.73	0.48	0.52
NT	2,557	288	8.88	0.55	0.45
PA	22,086	2,545	8.68	0.56	0.44
SP	12,020	1,478	8.13	0.57	0.43
ST	22	4	4.95	0.69	0.31
US	12,666	1,357	9.33	0.54	0.46

B.2.2 Regional Weights

Incorporating Benefits – To enhance the national-level weighting scheme derived from the EBI, benefits data are used to develop regional weights that reflect the value of environmental benefits generated under these programs. Earlier benefits studies (Feather and Hellerstein, 1997; Feather et al., 1999) examined the value of reducing soil erosion through the CRP across the United States. These studies accounted for the variable effect of water quality (soil erosion) on recreational expenditures from the National Survey of Recreation and the Environment and estimated the marginal benefit of reducing soil erosion by 1 ton for recreational uses at each of the NRI survey points. Individual estimates were imputed to the national population using a calibrated benefits transfer approach (Feather and Hellerstein, 1997).

Following the benefits transfer, the marginal benefit of reducing a ton of erosion in any particular region can be generated for each Farm Production Region. Based on simulation estimates of actual soil erosion occurring in these regions, a weighted average for valuing soil erosion at the Farm Production Region level is determined (table B.8). Implicit in these regional weights will be the population size and characteristics embodied in the benefits transfer exercise conducted by Feather et al. (1999). Values show that the average marginal benefit of reducing soil erosion is closely linked to population density per square mile.

Table B.8. Marginal benefits of reducing soil erosion

Region	Mean marginal benefits $ per ton	Weight %	Population density per sq. mile
AP	1.81	0.09	137.92
CB	1.91	0.09	149.21
DL	1.32	0.07	70.06
LA	2.13	0.11	106.02
MN	1.57	0.08	21.23
NP	0.50	0.02	19.09
NT	4.64	0.23	346.12
PA	2.61	0.13	135.57
SP	2.21	0.11	73.51
ST	1.47	0.07	169.25

Source: Feather et al. (1999).

This regional measure of the value of environmental benefits (i.e., marginal benefits of reducing soil erosion) is multiplied by the national-level weights developed using the EBI (table B.2) to determine one set of regional weights for weighting multiple environmental criteria (table B.9).

Table B.9. Benefit-adjusted regional weights using the EBI

	Regional weights									
Resource concerns	AP	CB	DL	LA	MN	NP	NT	PA	SP	ST
Carbon emissions	0.12	0.12	0.09	0.14	0.10	0.03	0.30	0.17	0.14	0.09
Wind erosion	0.12	0.12	0.09	0.14	0.10	0.03	0.30	0.17	0.14	0.09
Productivity loss	0.39	0.41	0.28	0.46	0.34	0.11	1.00	0.56	0.48	0.32
Pesticide leaching	0.03	0.03	0.02	0.04	0.03	0.01	0.08	0.05	0.04	0.03
Nitrogen leaching	0.06	0.06	0.04	0.07	0.05	0.02	0.16	0.09	0.07	0.05
Pesticide runoff	0.01	0.01	0.01	0.01	0.01	0.00	0.03	0.02	0.02	0.01
Sediment runoff	0.09	0.10	0.07	0.11	0.08	0.02	0.23	0.13	0.11	0.07
Nitrogen runoff	0.04	0.04	0.03	0.05	0.03	0.01	0.10	0.06	0.05	0.03
Phosphorus runoff	0.04	0.04	0.03	0.05	0.03	0.01	0.10	0.06	0.05	0.03

This assumes that the region with the highest average marginal benefit for reducing soil erosion (as measured by recreational benefits) is also the region with the highest value of overall environmental quality.

A similar set of regional weights can be developed from the EQIP contract data (table B.4) following the same procedure as for the national-level weights (table B.10).

Table B.10. Regional weights using EQIP contract data

	Regional weights									
Resource concerns	AP	CB	DL	LA	MN	NP	NT	PA	SP	ST
Carbon emissions	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Wind erosion	0.01	0.00	0.00	0.02	0.10	0.18	0.03	0.08	0.02	0.00
Productivity loss	0.43	0.48	0.62	0.52	0.61	0.51	0.22	0.35	0.46	0.41
Pesticide leaching	0.07	0.05	0.07	0.06	0.01	0.01	0.06	0.08	0.00	0.02
Nitrogen leaching	0.13	0.10	0.05	0.13	0.13	0.07	0.20	0.07	0.03	0.16
Pesticide runoff	0.00	0.02	0.00	0.02	0.00	0.03	0.06	0.11	0.04	0.02
Sediment runoff	0.31	0.16	0.22	0.10	0.14	0.10	0.13	0.10	0.27	0.05
Nitrogen runoff	0.03	0.10	0.01	0.08	0.00	0.04	0.17	0.11	0.10	0.23
Phosphorus runoff	0.02	0.09	0.02	0.08	0.00	0.05	0.14	0.09	0.07	0.10

After an initial comparison of the weights in tables B.9 and B.10, one might expect a working-land program using regional weights derived from the EBI to result in greater levels of carbon sequestration than one using weights derived from past EQIP expenditures. Similarly, a program using the regional weights from the EBI might be expected to enhance environmental performance of working lands to a greater degree in the Northeast than a program using regional weights derived from EQIP expenditures. However, these weights do not reflect the extent to which practices that address one resource concern are complements or substitutes for other resource concerns. For example, various tillage practices that reduce sediment runoff may enhance soil productivity, but might result in increased pesticide and nitrogen leaching. Hence, we cannot say, prior to empirically simulating these weighting schemes within the framework of alternative working-land programs, what the potential results may be.

Additional Weighting Schemes – In addition to the two regional-level weighting schemes described above, two other weighting schemes were considered at the regional level. The first of these are regional weights derived from the EBI, similar to table B.9, but with zero weights for changes in soil productivity (table B.11).

Table B. 11. Regional weights derived from the EBI without valuing soil productivity

	Regional weights									
Resource concerns	AP	CB	DL	LA	MN	NP	NT	PA	SP	ST
Carbon emissions	0.21	0.22	0.15	0.24	0.18	0.06	0.53	0.30	0.25	0.17
Wind erosion	0.21	0.22	0.15	0.24	0.18	0.06	0.53	0.30	0.25	0.17
Productivity loss	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Pesticide leaching	0.03	0.03	0.02	0.04	0.03	0.01	0.08	0.05	0.04	0.03
Nitrogen leaching	0.14	0.15	0.10	0.17	0.12	0.04	0.36	0.20	0.17	0.11
Pesticide runoff	0.02	0.02	0.02	0.03	0.02	0.01	0.06	0.03	0.03	0.02
Sediment runoff	0.16	0.17	0.12	0.19	0.14	0.04	0.41	0.23	0.19	0.13
Nitrogen runoff	0.07	0.07	0.05	0.08	0.06	0.02	0.18	0.10	0.09	0.06
Phosphorus runoff	0.06	0.06	0.04	0.07	0.05	0.02	0.15	0.09	0.07	0.05

A common feature of both the EBI- and EQIP-derived weights is the relatively large weight that CRP and EQIP place on maintaining soil productivity. Such large weights may be unnecessary, as producers likely prefer practices that maintain or improve soil productivity, which has inherent value. Producers would be expected to directly benefit from enhancing soil productivity, and therefore, additional payments are not necessary to encourage such behavior. Consequently, the weights on the other indicators are augmented relative to their initial importance in the weighting scheme.

Given the amount of information necessary to generate weights for multiple environmental criteria, program decisionmakers may prefer to simplify this procedure and weight practices that address alternative resource concerns equally across all the nine environmental indicators and 10 regions (i.e., a weight of 0.11 for each resource concern in each region). As mentioned earlier, a uniform weighting scheme implicitly adopts the point equivalency ratios (eq B.3) belying the 0-1 benefit index (I_{kji}) for each pollutant (j). That is to say, the practice that potentially discharges the largest amount of phosphorus to surface waters is weighted equivalently to the practice that potentially sequesters the least amount of carbon, etc.

Web Appendix C
Conservation Benefits, Installation Costs, and Land Rental Rates

The data used to analyze correlation among conservation benefits, conservation costs, and agricultural land rental rates come from a number of sources including:

- The Natural Resource Conservation Service (NRCS) Work Load Assessment (WLA);
- Environmental Quality Incentives Program (EQIP) contract data;
- National Resource Inventory (NRI) point data files;
- 1997 Census of Agriculture;
- An ERS database of estimates – drawn from previous studies – of a wide range of benefits that are likely to flow from soil erosion reduction and the wildlife benefits of establishing conservation cover from partial field practices such as grassed waterways and filter strips;
- Rental rate data developed for the Conservation Reserve Program (CRP) and Grassland Reserve Program (GRP).

C.1 Resource Concerns

Data on acres that require the application of one or more conservation practices to address a resource concern is from the WLA. For each county, the WLA provides the acreage of various land types (e.g., cropland, pasture) that require the application of practices for various resource concerns. A total of 573 million acres of cropland and grazing land have some treatable resource concern (Table C.1).

Table C.1. Summary of WLA data on resource concerns by land type and resource concern

	Resource concerns					
Land type	Soil erosion	Nutrient & pest mgmt	Irrigation water	Grazing	Wildlife	Totals
			million acres			
Cropland	162	35.8	42.6	1.4	6	247.8
Grazing land	55.8	12.9	2.1	235.6	18.8	325.2
Totals	217.8	48.7	44.7	237	24.8	573

To adapt WLA data for use in comparing benefits and costs, several adjustments were necessary. First, separate estimates of wind and water erosion concerns were needed. This report assumes that the proportion of acres needing treatment for wind erosion is roughly equal to the proportion of acres with wind erosion in excess of the soil loss tolerance, or "T" level. A similar procedure is used to determine the number of acres that were assigned a water erosion concern. Data on wind and water erosion is from the 1997 NRI. To allocate other resource concerns among non-irrigated and irrigated cropland, it is assumed that resource concerns are distributed proportionately among irrigated and non-irrigated cropland, by county. Data on irrigated and nonirrigated cropland is obtained from the 1997 Census of Agriculture.

C.2 Conservation Practice Installation Costs

Conservation practice installation costs are estimated from EQIP data for 1996-2001. The cost of addressing a given resource concern is the average cost of installing or adopting practices that are typically used to address it. Practices are grouped according to the physical processes they affect, i.e., practices that reduce water erosion are grouped together, etc. Groupings are similar to those used in *Environmental Quality Incentives Program: Benefit-Cost Analysis* (USDA, NRCS, 2003). To address a resource concern, producers would be required to address one or more of these physical processes. The average per-acre cost of practices used to address various physical effects is calculated from a subset of 33 of the practices most frequently used in EQIP contracts.

To estimate the average cost of installing or adopting conservation practices used to address specific resource concerns, total practice cost is used. For structural practices, total cost is the cost-share paid divided by the cost-share rate. For management practices, total cost is estimated as the maximum allowed incentive payments, obtained by dividing payment amount by the proportion of the maximum that is actually paid to the producer. While the maximum payment rates are designed by NRCS to approximate local costs, there remains considerable uncertainty about the actual costs of applying management practices. Nonetheless, these rates are the best available proxy for the cost of applying management practices.

For some practices, the extent of application is described in units other than acres. For example, the extent of terraces cost-shared is described in terms of linear feet. For these practices, conversion factors developed for the EQIP benefit-cost analysis are used to convert units into acres treated.

Although the data are identified to counties, NASS Agricultural Statistics Districts (ASDs) were used as the basic unit for averaging costs. Historically, EQIP has not been a large program and many counties include only a small number of EQIP contracts. Thus, a larger, multicounty area is likely to provide more reliable estimates of practice installation cost while also capturing spatial variation in conservation costs. ASDs were selected for this purpose because they are sub-State areas defined along county lines. Within each ASD, the average cost of practices addressing specific resource concerns is the acre-weighted sum of practices generally used to address the resource concern.

C.3 Benefits of Conservation

The benefits generated by the application of conservation practices are estimated using benefits transfer techniques. Benefit estimates were drawn from the literature and applied using additional data and physical process models. For example, water quality benefits are typically expressed in terms of damage reduction per ton of soil erosion reduction. These benefits can be applied on a per-acre basis using estimates of potential erosion reduction derived from the NRI.

C.3.1 Water Quality

Control of water erosion can improve water quality. Benefits generally grouped under the rubric "water quality" actually represent a wide range of distinct benefits, including water-based recreation, loss of reservoir storage capacity due to silt buildup, dredging costs for navigation, and additional water treatment costs for both drinking and industrial use. Increased benefits to water-based recreation from reduced soil erosion are based on estimates by Feather and Hellerstein (1997). Hansen et al. (2002) estimate the cost of soil erosion based on the cost of downstream dredging to maintain navigation channels. Other benefits are based on Ribaudo (1990).

Benefit estimates from these studies are in dollars per ton of soil conserved. To convert these figures to dollars per acre, likely water erosion reductions were estimated using historical data from NRI. Within a watershed (8-digit hydrologic cataloguing unit), expected erosion reduction due to practice application is estimated as the acre-weighted average erosion reduction on NRI points where: (1) erosion was above the soil loss tolerance (T) level in 1992; (2) erosion was reduced by 25 percent or more between 1992 and 1997; and (3) the erosion rate was below 1.25*T in 1997.[13] The same procedure is used to estimate erosion reductions for both cropland and grazing land.

C.3.2 Air Quality

Control of wind erosion can improve air quality. Benefits generally grouped under the rubric "air quality" include, among other things, decreased cleaning costs due to dust accumulation and health effects. Like water benefits, data is provided on the basis of benefits per ton of soil conserved. These benefit estimates are converted to a per-acre basis using a procedure analogous to that outlined above for water erosion. Ribaudo et al. (1990) developed regional measures of the cost of particulate pollution caused by wind erosion. The cost model is estimated using contingent valuation techniques and data from a survey of households in New Mexico (Huszar and Piper, 1986). Benefit estimates are provided per ton of soil conserved. Per-ton estimates are converted to a per-acre basis using procedures analogous to those used for water erosion.

C.3.3 Soil Productivity

Conservation of soil depth preserves soil productivity. Soils can also lose productivity, in the short run, when nutrient or other costly production inputs are lost with the soil. Reductions in soil erosion will increase the future productivity of farmland and reduce the loss of soil nutrients that can be washed away with the soil. For this study, average losses in soil productivity and nutrients per ton of soil erosion are derived from Ribaudo et al. (1990).

C.3.4 Wildlife Habitat

Benefits used for the calculations in this report are based on an ERS study described in Feather et al. (1999). Benefits are based on *use values*, or the value derived from directly using the resource – specifically for wildlife viewing and

[13] The factor of 1.25 accounts for the soil-erosion tolerance allowed producers.

pheasant hunting. Although improvements in wildlife habitat benefit a number of avian species, the demand for pheasant hunting was easier to quantify based on existing recreational data. The ERS model evaluates the quantity and quality of the cover available for specific avian species, then estimates the surplus resulting from enrolling land in CRP. Since establishing grassland or forest cover creates suitable habitat for birds, small game, and large game, hunters and wildlife viewers then benefit from these increased populations. The model also incorporates travel costs, landscape diversity, and population density. There are limitations associated with using benefits estimated for the CRP in the context of a working-land program. However, most of the practices that generate wildlife benefits in the working-land context produce wildlife cover similar to that found on CRP land. Grassed waterways, windbreaks, and similar practices generate wildlife benefits in much the same way CRP would. Nonetheless, this report addresses any difference by reducing the wildlife benefits estimated to be generated through CRP by 50 percent.