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THE OFF-SITE ECONOMIC BENEFITS OF SOIL CONSERVATION: A REVIEW AND DISCUSSION OF RECENT LITERATURE ON THE RECREATIONAL DEMAND FOR WATER QUALITY IMPROVEMENT

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INTRODUCTION

The erosion of agricultural lands leads to two types of economic damages. The first is the on-farm loss of soil productivity, which is a consequence of the loss of soil structure, nutrients, organic matter, micro-organisms, and soil moisture-holding capacity. Productivity loss may initially be negligible, but will become acute as the root-depth threshold is approached. Losses are experienced as reduced yields, or increased input costs when synthetic inputs are used as substitutes for soil nutrients. The second category of losses or damages occurs off-site, and involves the pollution of surface waters by sediment, nutrients, and pesticides. Damages and losses are experienced through degradation of water-based recreational opportunities, increased municipal and industrial water treatment costs, accelerated loss of water storage capacity, aggradation of navigation channels, siltation of water conveyance channels, increased flood damages, and damage to aquatic ecosystems.[1]

Concern for soil erosion and strategies to conserve soil historically have focused on the first of these two categories. However, evidence has accumulated that agricultural nonpoint pollution contributes substantially to surface water quality problems throughout the nation. The Environmental Protection Agency (EPA) has estimated that agricultural nonpoint pollution significantly affects water quality in 68% of all drainage basins in the U.S., and almost 90% of those in the corn belt (Braden and Uchtmann, 1985). Clark (1984), reviewing several recent studies, estimates that the nonpoint-sources contribute as much as 73% of Biochemical Oxygen Demand (BOD), 99% of Total Suspended Solids (TSS), 88% of total Nitrogen (N), 94% of total Phosphorus (P), and 98% of bacteria in

U.S. waterways. A growing body of evidence suggests that the off-site economic damages associated with agricultural soil losses probably exceed the on-farm productivity losses in overall magnitude.[2] The benchmark study by Clark (1985) concludes that off-site damages nationwide attributable to soil erosion exceed $8 billion annually, with farmland's share being approximately $3 billion (1986 US $). Other national studies indicate that these figures are at least of the correct order of magnitude (AAEA, 1986).

Freeman (1984) distinguishes between "top-down" and "bottom-up" approaches to benefit estimation and transfer. In the former, benefits are estimated in an aggregate, such as the national level, and techniques are devised to disaggregate or allocate these estimates between regions or other subdivisions. If the subdivision is small enough, benefits (costs) are estimated locally, possibly on the individual or household level, and aggregated upward. National and regional studies of the economic benefits from water quality improvement, while effective in redirecting the attention of soil conservation policymakers, are of little value in providing estimates of economic benefits associated with individual soil conservation projects or practices. A realistic and defensible set of benefit guidelines for project analysis is required, however, if conservation resources are to be targeted effectively. These benefit guidelines must reflect the spatial variation in the characteristics of demand for water quality improvement, particularly given that targeting decisions themselves must be made in a regional or local context.

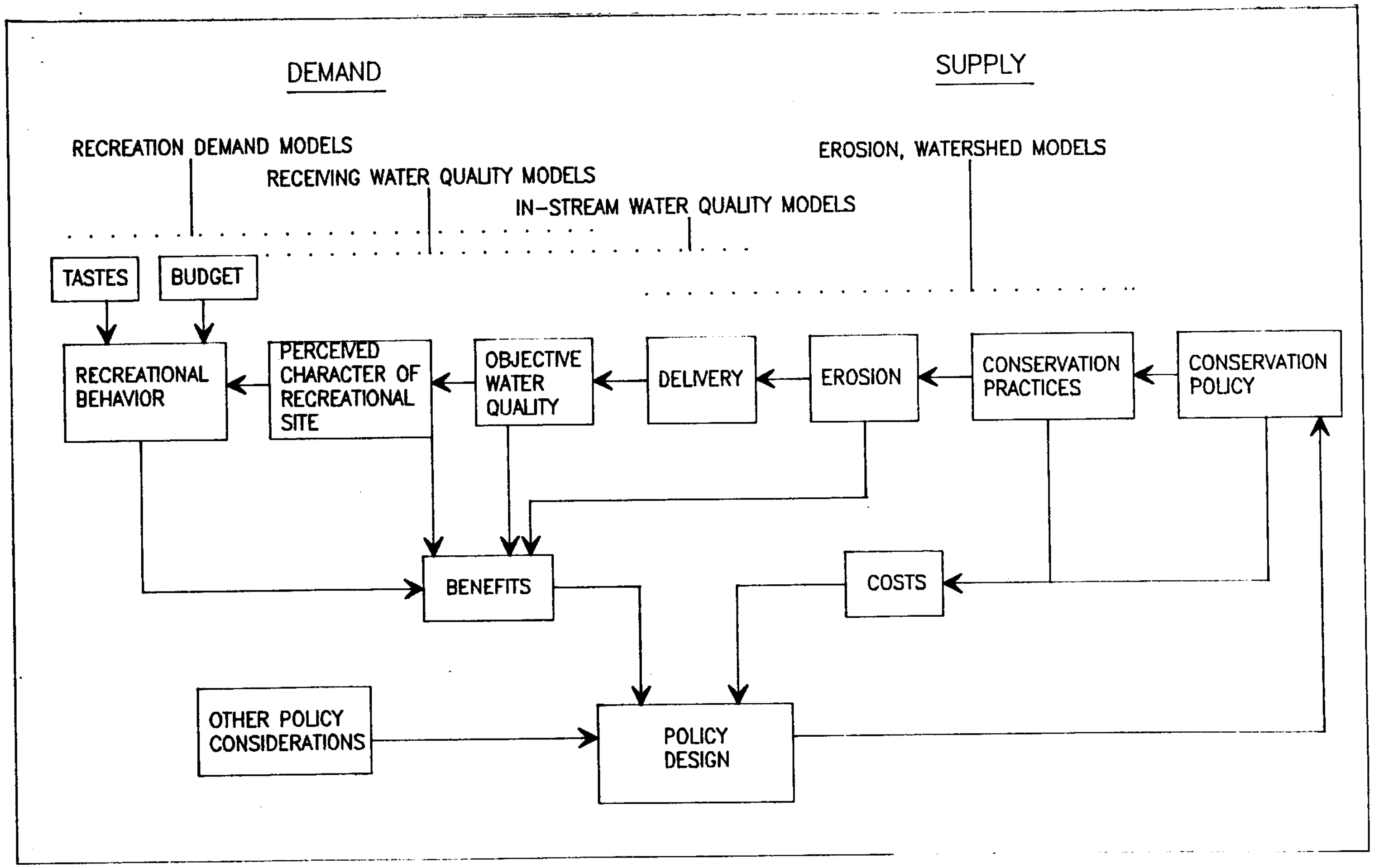
This report contains a review of empirical studies yielding estimates of the economic benefits of water quality improvement. The focus is on the demand for water quality improvement by recreational users. The purpose is to improve the understanding of how water quality benefits associated with recreation can be used in benefit-cost analysis and the development of criteria for "targeting" conservation resources at the Federal, State, or local levels. Existing studies of the national economic benefits from water quality improvement support the belief that recreational benefits are among the largest classes of such benefits, possibly the largest. Several of these studies will be summarized in the following section.

Recreational benefits are but one of several interrelated objectives of soil conservation policy. Between the design and implementation of policy and the subsequent realization of benefits lies a system of intermediate linkages. Each linkage involves a unique scientific discourse and modeling approach, and each introduces a degree of uncertainty derived either from the stochastic behavior of natural systems or the idiosyncratic nature of human behavior:

> "Policymakers cannot affect water quality directly; they can only affect the decisions made by individuals. These decisions determine the actual conservation measures used. The effect of the measures taken on water quality will depend on a highly complex set of physical relationships among erosion rate, sediment transport, and a variety of in-stream, hydrologic characteristics that ultimately affect water quality. There are additional ecological considerations that then relate changes in water quality to changes in the ecological system and aquatic habitat. Finally, there are the complex human factors (preferences and decisions) that provide a value to the various physical factors" (Fletcher, 1987).

Figure 1 depicts the process linking conservation policy implementation to the realization of recreational benefits from improved water quality. The focus of this document is on the final linkage,

Figure 1 – Process Relating Conservation Policy to Water Quality Benefits



between perceived site characteristics such as water quality and recreational behavior (see Rodgers, 1989, for a review of the literature on each of the linkages identified in Figure 1).

BENEFITS OF WATER POLLUTION CONTROL: MACRO STUDIES

Passage of the Federal Water Pollution Control Act (WPCA, 1972) amended as the Clean Water Act (CWA, 1977) stimulated interest in the measurement of the national economic benefits associated with the achievement of mandated water quality standards. The Acts are regulatory in approach, and are primarily concerned with point sources of pollution. Section 208 authorizes the development and implementation of area-wide waste treatment plans to manage and reduce nonpoint-source pollutants, including those associated with agriculture, but cropland nonpoint sources in practice have not been subject to regulation under the Act. It is reasonable to conclude that for this reason, and due to the limited availability of data on nonpoint pollution in the early 1970s, the national studies that estimated benefits emphasized the reduction of point-source pollution (Tables 1 and 2).

A 1978 EPA-sponsored study (EPA I) estimated the national water pollution damages from all artificial sources in 1973. These were considered to be equivalent to the benefits that would result from reducing all human-source pollution to the threshold at which damages can be observed. The estimates of total damages (converted to 1986 dollars using the GNP Deflator) ranged from $10.5 billion to $43.5 billion, reflecting substantial uncertainty. The largest damages and greatest

Table 1 - Estimates of National Benefits from Water Pollution Abatement.
(Billions of 1986 $)

Category of Benefit	EPA I (a)	% of Total Benefits	EPA II (b)	% of Total Benefits
Recreation:	14.6	62.4%	3.9	30.3%
Aesthetic, Ecological:	3.5	14.9%	8.0	62.1%
Health:	1.4	5.9%	0.004	0.0%
Materials Damage and Production:	3.9	16.8%	1.0	7.6%
Total:	23.5	100.0%	12.8	100.0%

(a) Benefits in 1973 from total abatement of water pollution from human sources. Source: EPA, (1978), p. I 28.

(b) Benefits in 1975 from meeting 1977 objectives of 1972 WPCA. Source: Feenberg and Mills, (1980), p. 153.

Table 2 - Estimates of National Benefits from Water Pollution Abatement. (Billions of 1986 $)

Category of Benefit	Freeman (1979) (a)	% of Total Benefits	Freeman (1982) (b)	% of Total Benefits
Recreation:	11.1	54.7%	7.3	48.9%
Nonuser Benefits: Aesthetics, Ecology, and Property Value	3.4	16.5%	1.9	12.8%
Commercial Fisheries:	1.3	6.5%	1.3	8.5%
Diversionary Uses:				
Drinking Water-Health	1.6	7.9%	1.6	10.6%
Municipal Treatment	1.5	7.2%	1.4	9.6%
Households	0.4	2.2%	0.5	3.2%
Industrial Supplies	1.0	5.0%	1.0	6.4%
Total:	20.3	100.0%	15.0	100.0%

(a) 1978 benefits from meeting 1985 objectives of 1977 CWA. Source: Vaughan, and Russell (1982), p. 7.

(b) Annual benefits from meeting 1985 objectives of 1977 CWA. Source: A. M. Freeman III (1982), p. 170.

degree of uncertainty regarding them are associated with outdoor recreation (EPA, 1978).

A second EPA-sponsored study (EPA II) estimated the national benefits associated with meeting the 1977 CWA goals using similar methodology (Freenberg and Mills, 1980). The lower figures (EPA II in Table 1) reflect, in part, the fact that water pollution control was required to follow a phased implementation of increasingly strict quality standards, taking effect in 1977, 1983, and 1985. The 1977 standards were considerably less stringent than those implicitly assumed by EPA I. Recreation, aesthetic, and ecological benefits collectively represent over 90% of projected benefits in EPA II.

Freeman (1979) analyzed the results of several existing studies to obtain an estimate of national benefits from meeting the 1985 "Best Available Technology" (BAT) water quality objectives of CWA. The 1985 BAT standards approximate the threshold levels assumed in EPA I. His total 1978 point estimate of $20.3 billion is also in good agreement with the $23.5 billion point estimate obtained for EPA I (both in 1986 $). Similarly, Freeman attributed 55% of benefits to recreation, which is reasonably close to the 62% estimated in EPA I.

Freeman (1982) proposed a revised set of benefit estimates, again assuming the successful implementation of 1985 BAT water quality standards nationwide. The revisions are based on the preliminary findings of Vaughan and Russell (1982), who developed a modeling approach allowing improved geographical coverage and regional detail. Freeman's 1982 estimates are lower than his 1979 figures, although both the overall magnitude of benefits and the percentage attributable to recreation support the general consensus that recreational benefits from water quality improvement are significant in both relative and absolute terms.

Clark II, et al. (1985) have estimated the off-site damage costs of soil erosion, and more specifically those due to agriculture, at the national level. To obtain their estimates, Clark and colleagues relied on both the national estimates of damages to recreational and commercial fishing from all sources of water pollution cited above (Freeman, 1982; Vaughan and Russell, 1982), and on primary, site-specific engineering and economic studies of reservoir and channel dredging and excavating, water treatment costs, flood damages, and other categories of damage related to sediment and nutrient pollution of surface water. Figures were aggregated, and in some cases extrapolated, to obtain both ranges and single-value estimates of off-site damages nationwide. The authors further assumed that, for most categories of damage, one-third of sediment and associated pollutants originated on agricultural land, and a corresponding fraction of damages would thereby be attributable to agriculture (see Table 3).[3] A comparable set of national damage estimates from erosion were prepared by the AAEA Task Force (1986). Most of their estimates were based directly on Clark's figures, or were obtained using methodology and sources similar to Clark, which explains the nearly identical estimates in most categories (see Table 4).

Further evidence of the importance of water-based recreation is provided by the assessment of economic benefits from 28 projects conducted under Sec. 314 of the Clean Lakes Program (1980) (EPA, 1980). Between 1976 and 1979, the EPA awarded 105 project grants in 37 states totalling $40 million (1980 $). Benefits estimated for these 28 projects fall into

Table 3 - Summary of Estimated National Off-Site Damages Due to Erosion.

Type of Impact	Range of Estimates: Low	High	Single Value Estimate	Cropland's Share
	Millions of 1986 $			
In-Stream Effects:				
Biological Impacts:	(no estimates of aesthetic or ecological impacts)			
Recreation:	1,275	7,515	2,684	1,114
Water Storage:	416	2,147	926	295
Navigation:	564	1,074	751	242
Other In-Stream Uses:	617	3,355	1,208	429
Total In-Stream:	2,872	14,090	5,569	2,080
Off-Stream Effects:				
Flood Damages:	590	1,744	1,033	335
Water Conveyance:	188	403	268	134
Water Treatment:	67	671	134	40
Other Off-Stream Uses:	537	1,235	1,074	376
Total Off-Stream:	1,382	4,053	2,509	886
Total All Effects:	4,254	18,142	8,078	2,966

Source: Clark, et al., (1985), p. 175. Figures are for an arbitrary recent year.

Table 4 - Summary of Estimated National Off-Site Damages Due to Erosion.

Type of Impact	Range of Estimates: Low	High	Single Value Estimate	Cropland's Share
	-----------Millions of 1986 $----------			
In-Stream Effects:				
Biological Impacts:(a)				
Recreation:	1,275	7,515	2,818	1,114
Water Storage:	671	1,744	1,087	349
Navigation:	564	1,074	751	242
Other In-Stream Uses:	564	3,757	1,114	443
Total In-Stream:	3,073	14,090	5,770	2,147
Off-Stream Effects:				
Flood Damages:	658	1,879	1,033	335
Water Conveyance:	188	403	268	134
Water Treatment:	67	671	134	40
Other Off-Stream Uses:	-121	-496	-174	-54
Total Off-Stream:	792	2,456	1,261	456
Total All Effects:	3,865	16,546	7,032	2,603

Source: AAEA, p. 39.

(a) No estimates were made of aesthetic or ecological impacts.

12 categories: recreation, aesthetics, flood control, economic development, pollution reduction, and miscellaneous items including resource recovery and reduced management costs. The present discounted value of benefits which could be quantified was $127.5 million (1980 $). Recreation benefits are the most prevalent category, projected to be significant for 25 of the 28 projects and the largest category of benefits for 20 of the 28 (Table 5).

TARGETING CRITERIA TO MEET MULTIPLE OBJECTIVES

One justification for quantifying the benefits of site-specific water quality improvement lies in their value as criteria for targeting soil conservation resources. Within the context of benefit-cost analysis, targeting serves as a means of establishing the most efficient allocation of public resources; i.e., of minimizing the cost of achieving a prespecified policy goal or goals.[4] Targeting has been an explicit component of USDA soil conservation programs since the 1977 RCA, and even more so since the National Conservation Program was enacted in 1982.

Tinbergen (1952), established the conditions under which policy instruments are likely to succeed. He states that "...for each policy objective there should be at least one instrument, and each instrument should be carefully designed to have maximum impact on its primary objective." At least four distinct policy objectives can be identified for combined USDA soil conservation programs, and other Federal programs concerned with land use and environmental quality. These are: (1) protection of the productivity of the soil base, (2) preservation and improvement of water quality, (3) economic and social welfare of farmers

Table 5 - Summary of Benefit Estimates for 28 Clean Lakes Projects. (a)

Benefit Category >	Recreation, Aesthetics, Fish & Wild.	Flood Control, Economic Development	Agriculture	Property Values	Public Health, Water Supply	Miscellaneous	Total Annual Benefits
Lakes:				1986 $ (b)			
Annabessacook	(c) +	123,445	+	29,762,380	0	0	29,885,825
Bomoseen	81,543	0	0	0	0	(13,302)	68,240
Buckingham	24,343	+	0	0	0	0	24,343
Charles	406,782	0	0	0	0	0	406,782
Clear	46,691	+	0	0	+	0	46,691
Cochrane	5,321	0	0	+	0	0	5,321
Collins Park	9,844	0	0	0	0	0	9,844
Ellis	677,083	48,952	1,330,222	0	+	62,520	2,118,777
59th Street Pond	921,844	+	0	0	+	0	921,844
Frank Holton	354,903	+	0	0	+	+	354,903
Henry	5,321	0	+	+	0	136,082	141,403
Jackson	313,799	0	0	+	0	0	313,799
Lansing	220,285	+	0	+	0	0	220,285
Liberty	140,737	0	0	+	+	0	140,737
Lilly	+	0	0	3,831,039	0	0	3,831,039
Little Pond	+	0	0	0	40,439	0	40,439
Loch Raven	162,819	0	0	0	2,031,116	0	2,193,935
Medical	118,922	0	0	299,300	0	0	418,222
Mirror, Shadow	59,594	0	0	+	+	0	59,594
Moses	101,895	0	0	0	0	+	101,895
Nutting	591,151	12,903	0	+	+	0	604,054
Penn	29,132	+	0	0	0	44,030	73,162
Rivenna	4,789	0	0	0	165,480	0	170,268
Steinmetz	24,077	+	0	0	0	0	24,077
Temescal	255,270	0	0	0	4,922	94,180	354,371
Tivoli	45,760	+	0	0	+	0	45,760
Vancouver	6,168,238	+	19,950,000 (d)	0	+	0	6,168,238
Washington Park	23,013	0	0	0	0	0	23,013

(a) Source: EPA (1980)
(b) 1980 figures converted to 1986 $ using GNP Deflator
(c) + Indicates expectation of probable benefits, but no estimate.
(d) One-time benefit, obtained by valuing dredge spoil at $2.00 per cubic yard for agricultural use.

and rural communities, and (4) overall economic efficiency. Related objectives, or constraints within which policy must be formulated, include "fairness" (avoidance of policies that place a disproportionate share of adverse economic consequences on specific parties), and consistency or compatibility with other USDA programs and policies, such as supply control. It is, therefore, unlikely that any simple targeting criterion will be found which adequately addresses all existing policy objectives. In addition, there is reason to believe that currently employed targeting criteria can be improved upon.

From the beginning of the Federal involvement in soil conservation, various targeting criteria have been proposed or adopted as guidelines. The first of these was gross erosion (tons of soil dislocated per unit area), typically measured in tons per acre per year (TAY). Gross erosion is effectively estimated using the Universal Soil Loss Equation (USLE), which predicts gross sediment yield as a function of the following location-specific physical parameters: rainfall intensity (R), soil erodibility (K), field length (L) and slope (S), and of cropping (C) and management (P) practices (Wischmeier and Smith, 1978). Although gross erosion is appealing as a targeting criterion, it provides limited information on the location-specific relationship between soil loss and productivity loss. Specifically, it does not reflect the existing soil profile, depth, erosion history, and gross rate of soil formation. In addition, gross erosion is not necessarily correlated with sediment and nutrient delivery to surface waters in any straightforward manner, and so may provide inadequate information on the relative contributions of land units to nonpoint pollution loadings and water quality problems.

Ribaudo (1986a) provides empirical evidence for the inadequacy of gross erosion as a multi-objective targeting criterion. He classified and ranked 99 in the Water Resource's Council Aggregated Sub-Areas (ASA's), both by gross erosion rates and by relative contribution to surface water pollution. Most of the ASA's with the highest gross erosion were located in the corn belt. The ASA's were also ranked for their potential off-site damage on the basis of (1) ambient levels of three pollutants associated with agriculture (TSS, N, P), (2) agriculture's relative contribution to water quality problems, and (3) instream recreational water use and withdrawal levels. After comparing the two sets of rankings, Ribaudo concluded that the sole use of on-site erosion criteria for program targeting will identify only a few of the regions which are important in terms of off-site benefits.

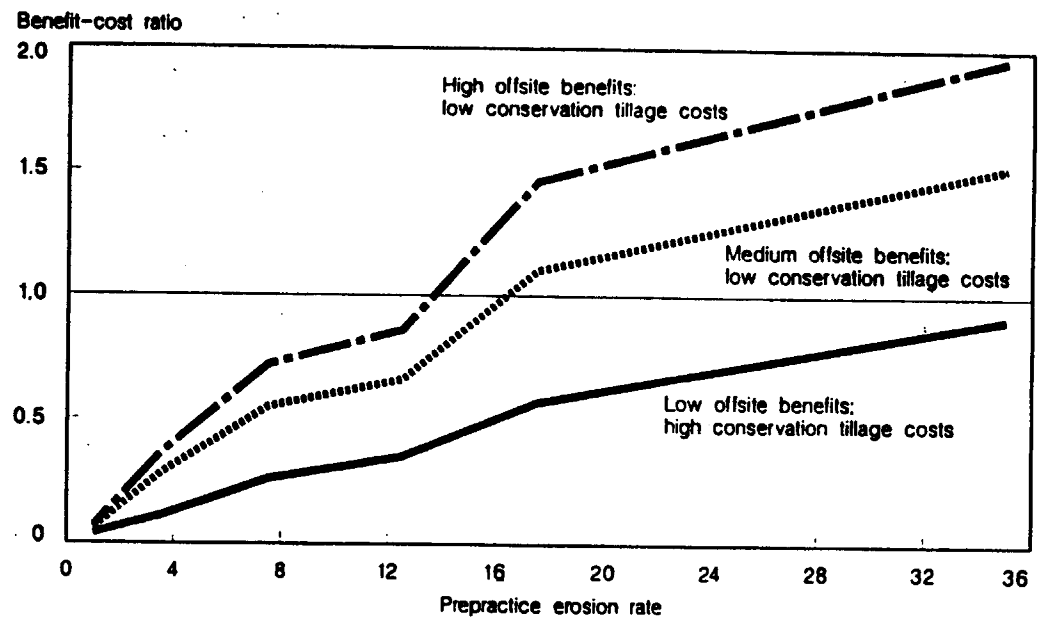
Since 1962, soil loss tolerance, or T-values, have been the preferred guideline for targeting conservation resources. T-values are location-specific and are defined as the "maximum level of soil erosion that will permit a high level of crop productivity to be sustained economically and indefinitely" (Wischmeier and Smith, 1978, p. 2). T-values have been established for most US soils by USDA soil scientists and agronomists on the basis of existing soil depth and productivity, location-specific physical factors such as rainfall and slope, and on the estimated rate of topsoil formation over time. T-values typically range between 2 and 5 TAY. They are used to target land units for conservation treatment by comparing gross erosion as predicted by the USLE to the established T-value. If gross erosion exceeds T, management or cropping systems that reduce erosion to less than T are recommended.

While clearly an improvement over gross erosion as a targeting criteria, T-values nevertheless have been criticized as inadequate for multi-objective planning. More specifically, they (a) reflect the rate of topsoil formation but fail to reflect differential rates of overall soil formation from parent material, (b) focus on soil depth rather than productivity per se, (c) are physical rather than economic measures, making them difficult to apply within benefit-cost analysis, and (d) say little about delivery of sediment and pollutants to surface waters, which limits their value where off-site damages are important (Strohbehn, 1986).

The final issue is particularly relevant, given the importance of benefit-cost analysis in Federal programs. A 1986 USDA evaluation of the three principal USDA conservation programs (CTA, GPCP, ACP) found that social benefit-cost ratios for these programs were highly sensitive to pre-treatment erosion rates, the relative magnitude of potential off-site benefits and the costs of conservation tillage (Strohbehn, 1986). These relationships are summarized in Figure 2.

A number of alternative targeting criteria have been proposed, reflecting dissatisfaction with T-values. They can be approximately subdivided into (a) on-site and productivity-related criteria; (b) criteria identifying potentially significant sources of surface water pollution loadings; (c) voluntary or market-driven approaches, and (d) integrated modeling approaches or combined damage functions. In evaluating the merits of any proposed targeting criteria, one must consider not only the specific policy objectives and the expected efficacy of the instrument in meeting them, but also the costs of information, implementation, and possibly enforcement as well.

Figure 2 - Erosion Control B/C Ratios Under Alternative Assumptions:



Source: Strohbehn (1986) p. 29

Bills and Heimlich (1984) advocate an approach to targeting that distinguishes between the physical and management-related components of erosion. Priority areas are those where (1) erosion significantly affects productivity <u>and</u> (2) erosion is controllable. Physical and management components of potential erosion are estimated by partitioning the USLE into: i) soil- and location-specific physical factors (R,K,L,S), which are "management-neutral" and provide an estimate of reference soil loss which would occur under continuous clean-till fallow, and ii) management terms (C,P), which correct reference soil loss to specific cropping and management practices. The authors define four erosion classes as follows:

Erosion Class:	Criteria:
Non-Erosive:	RKLS < 7
Moderately Erosive:	
Managed Below Tolerance:	RKLS > 7; USLE < 5
Managed Above Tolerance:	7 < RKLS < 50; USLE > 5
Highly Erosive:	RKLS > 50; USLE > 5

Data from the 1977 National Resource Inventory (NRI) indicates that 36.9% of U.S. cropland falls into the first category, 40.4% in the second, 14.9% in the third, and 7.8% in the Highly Erosive category. Land in the Non-Erosive category is unlikely to be vulnerable to significant productivity loss under any reasonable management scenario. Highly erosive land should be under permanent vegetative cover. Conservation resources should be targeted to moderately erosive lands, specifically those managed above tolerance and particularly those which are highly productive.

Runge, Larson, and Roloff (1986) developed a quantitative soil vulnerability measure (V-value) intended as a targeting criteria for conservation set-aside programs. First, the soil productivity index (PI) is defined as:

$$PI = \Sigma(A_i C_i D_i W)$$

for soil horizons $i = 1...n$
A_i is available water capacity sufficiency
C_i is bulk density sufficiency
D_i is pH sufficiency

The product is summed over the n soil horizons in the profile encompassing root depth zone. V is then defined as the rate of change in productivity with respect to changes in soil depth:

$$V = \Delta PI/\Delta d$$

d = soil depth in cm.

Then, an erosion rate, E_t, (in TAY) corresponding to any pre-specified change (reduction) in productivity over 100 years (ΔPI), e.g., 5%, can be calculated as:

$$E_t = [\Delta PI \cdot PI \cdot W]/V \cdot t$$

W = bulk density measure
t = time (100 years)

The resulting annual soil-loss threshold can be used as a targeting criterion, much as T-values, but it can more accurately capturing the localized relationship between soil depth and long-term productivity.

Maas, et al. (1985), viewed the protection of on-site soil productivity and the limitation of agriculturally induced water quality problems as conceptually distinct problems, each requiring a unique perspective ("land resources" vs. "water resources") and targeting guidelines. They further stress the importance of identifying the specific nature of the observed or perceived water quality problem (sediment, N, P, pathogens) as a precondition to selecting critical areas for treatment. And, along with Bills and Heimlich (1984), they emphasize the need to target areas that are characterized by <u>treatable</u> problems, i.e., where the application of BMP's can be expected to produce a meaningful improvement in water quality. A 9-step procedure is proposed: (1) identify location, type, and magnitude of water quality problem; (2) identify characteristics of the water resource: its potential and impaired uses; (3) estimate minimum reduction in pollution loading necessary to reverse impairment or restore water quality; (4) determine if this reduction can be realistically achieved through BMP's alone; (5) if yes, identify and rank the significant sources (locations) of pollution loadings (eg., gully erosion, animal confinement areas, heavily fertilized fields); (6) identify areas closest to affected watercourses, specifically within 1/4 mile: these are potentially critical areas; (7,8,9) eliminate from consideration sources either far upstream, under adequate management, or otherwise determined through on-site inspection not to represent critical areas.

Related decision rules have been proposed by Duda and Johnson (1985), Snell (1985) and Holstine and Lowman (1985). Duda and Johnson, drawing on studies conducted by the TVA on agricultural watersheds within western North Carolina, conclude that cost-effective water quality improvements can be obtained only through targeting agricultural pollution "hot spots," as opposed to critical watersheds, counties, or production regions. These hot spots are specific parcels which can usually be identified on the basis of existing information and are typically associated with "ephemeral gullies or agricultural activities near ditches and streams." Advantages of the proposed system include speed of implementation and low information-gathering costs.

Snell, seeking a less expensive and regionally scaled alternative to computer simulation, developed a map overlay technique, which allows identification of areas characterized by both high soil loss and high sediment delivery ratios. Soil loss is estimated using a graphic application of the USLE. Sediment delivery ratios are estimated using an 11-step flow chart that qualitatively sorts land units into high, medium, and low rates of field-to stream delivery. Data are encoded on 1:50,000 scale map transparencies, which are overlaid to allow rapid identification of areas with simultaneously high soil loss (> 11 T/Ha/Y) and high rates of delivery to surface waters.

Hostine and Lowman describe an allocation process used in Idaho which commences with the identification of sites experiencing severe erosion (> 5 TAY). These areas are linked to stream segments, which are then ranked qualitatively on three criteria: (1) present water quality, from EPA monitoring data; (2) affected population, including residents and nonresident recreationists; and (3) protection of high-quality waters. The factors are weighted 60%-30%-10%. The resulting ranking is categorical rather than quantitative: "first priority" and "second priority" problem areas are identified but not ranked within category.

All of the above, and many comparable decision guidelines recently appearing in the literature, can be characterized by (1) a reliance on existing land resource data rather than on primary data collection; (2) a sequential application of decision rules or sorting procedures, and (3) a heavy reliance on local expert opinion rather than large-scale modeling. Additionally, all emphasize speed of implementation.

Conservation Reserve Targeting

Voluntary land set-aside programs, notably the Conservation Reserve Program (CRP) of the 1985 Food Security Act, involve an implicit form of conservation targeting. The objectives of the CRP are both reduction of soil erosion and supply control, at minimum budget exposure. Eligible land must fall into the "highly erodible" category, defined as acreage having an Erodibility Index (EI) of 8 or higher, where EI is defined as (RKLS)/T. The program amounts to self-targeting in that farmers themselves select parcels to be idled from among their eligible acreage. The CRP is structured to reflect a commitment to fairness in that bids are evaluated within multi-county pools, insuring that Federal CRP rental payments are not concentrated in a particular region or regions characterized by high erosion rates. The competitive bid structure is designed to minimize program costs.

The CRP has, in its first three years of implementation, enrolled somewhat less acreage at a somewhat higher cost than originally projected. One predictable source of inefficiency, discussed by Taff and Runge (1987), results from a single policy instrument (the CRP) that is expected to serve at least three policy objectives: soil conservation, supply control, and budget discipline. This, and the fact that CRP effectively competes with another supply-oriented set-aside program (the Acreage Reduction Program, ARP) for set-aside acreage, leads to inefficiency in both programs.

Objectives are frustrated to a large extent because the most erosive lands are not necessarily the least productive, nor are the most productive acres the most resistant to erosion. Taft and Runge suggest a refinement in the eligibility criteria for both land set-aside programs that would increase the efficiency of both. Land would be cross-classified according to both the productivity index (PI and EI). The joint PI,EI rating would become the eligibility criteria for both set-aside programs. Acreage that is both productive and non-erosive would be ineligible for either program. Productive land that was erosive would be targeted for the ARP. Land that was both non-productive and erosive would be targeted for the CRP. Non-productive and non-erosive land would be exempt from either program, since the cultivation status of this land has little influence on either supply control or soil conservation.

Pierce (1987) extended this system of cross-classification to include a third dimension, the Nonpoint Source Index (NPSI). The NPSI is based on geographical position, e.g., proximity to surface water bodies, as well as on gross erosion potential. A high NPSI would indicate high potential contributions of sediment and nutrients to surface waters. The resulting three-dimensional policy targeting scheme based on PI, RI, and NPSI is reproduced below in Table 6.

Table 6 - Three-Dimensional Classification of Agricultural Land.

Productivity Index	Resistivity Index	NPS Pollution Index	Program Eligibility
Productive	Resistant	Low	None
" "	" "	High	Priority ARP
Productive	Non-Resistant	Low	Priority ARP
" "	" "	High	Priority CRP
Non-Productive	Resistant	Low	None
" "	" "	High	CRP
Non-Productive	Non-Resistant	Low	CRP
" "	" "	High	Priority CRP

Source: F.J. Pierce (1987), Figure 8.

While almost certainly achieving increases in economic efficiency and program effectiveness over existing criteria, the proposed targeting schemes nevertheless implicitly treat all potentially affected waters as possessing the same economic value. Efficiency could be further increased if targeting criteria reflected the differences in surface water value established around patterns of use and measurable demand.

Mathematical Models for Targeting

A final approach to targeting involves the use of mathematical models. These include regional, watershed, and farm-level models relating production, costs, and income to land features, cropping, and tillage systems; models relating productivity to soil depth and erosion; sediment dislocation and transport models; and regional water quality models. Increasingly, physical models are being linked to economic models, permitting optimization over both types of criteria. Examples include the CARD regional linear programming model linked with the Erosion-Productivity Impact Calculator (EPIC) and SOILEC (an economic model), used in the Resource Conservation Act (RCA) evaluation; the Watershed Evaluation Research System (WATERS), allowing watershed-level analysis of competing economic and environmental objectives; and the Agricultural Nonpoint-Source (AGNPS) model linked to economic models, (developed by the Department of Agricultural and Applied Economics at the University of Minnesota) for use in water quality targeting (Kozloff, 1989).

When adequately supported by the appropriate data, mathematical models are capable of identifying or targeting critical land units (fields, subwatersheds, regions) with a degree of accuracy meeting or exceeding that associated with most other methods. Constraints on the use of such models include availability and/or cost of required data, cost of acquiring and calibrating the models themselves, and current limitations in the accuracy of certain model components. The quantity and quality of land-base data has increased considerably as a result of the NRI and RCA. Models are increasingly designed or modified for use on micro-computers, lowering costs and increasing flexibility. Certain components of the modeling process, however, particularly river sediment transport, limnology, and recreational demand, continue to embody substantial uncertainty.

In summary: (a) many simple targeting criteria have been proposed which are adequate for single objectives, but which do not effectively address multiple objectives; (b) the more comprehensive and effective a set of criteria in meeting multiple objectives, the higher the likely information collection expense, and (c) even the most comprehensive targeting scheme may fail to achieve high economic efficiency in the use of conservation resources if the economic value or demand for surface water quality is not specified.

ESTIMATING BENEFITS OF WATER QUALITY IMPROVEMENT

The benefits associated with a water quality improvement is defined as the sum or aggregate of the monetary values assigned to the quality change by all individuals affected either directly or indirectly. Three aspects of this definition are of particular importance. First, benefits are experienced by _individuals_. Second, benefits are measured or aggregated across _all_ affected parties, a requirement necessary to distinguish economic benefits from economic impacts. Third, benefits are expressed in consistent, quantitative terms, specifically the "money metric" common to benefit-cost and other economic analysis.

A critical distinction must be made between the benefits of water quality improvement, and the damages that result from water quality degradation. In principle, one is the inverse of the other, differing only with regard to the reference point. Damages are experienced when water quality deteriorates from a clean state, actual or hypothetical, to a polluted state. Benefits, by contrast, are experienced when water quality is improved from a given degraded state, again actual or hypothetical, to a more desirable state. Damages should ideally be of the same magnitude as benefits, differing in sign only.

In practice, asymmetries exist that may cause the two measures to diverge. First is the irreversibility of investments or the inability to recover sunk costs. If a water treatment plant is initially designed to treat intake water high in sediment and dissolved solids, or if a reservoir has been designed with an enhanced sediment trap capacity, then it is highly unlikely that a future reduction in sediment load will lead to any reversal of these associated costs.

A second concern specific to water-based recreation is the phenomenon known as hysteresis. An aquatic ecosystem already damaged by nutrient enrichment may not respond immediately to a cessation in the discharge of pollutants, the recovery path may not retrace the trajectory of decline, and the post-abatement equilibrium may not resemble the pre-pollution equilibrium.

A related set of conceptual issues must be confronted in estimating the benefits or costs associated with a change in environmental quality. These concern the choice of an appropriate measure of welfare change, which is in part influenced by the property rights structure, and in part reflects the theoretical requirement that benefits reflect any changes in prices and real income resulting from the change. In describing an environmental quality change, two points of view are possible: the event is assumed to occur, or it is assumed to be avoided or foregone. In the first view, if the event is beneficial, a measure of the magnitude of the benefits to a given individual is the amount of money that could be taken from the individual to leave him or her precisely as well-off as before the event. This is the individual's "willingness to pay" (WTP) for the event. If, on the other hand, the change is detrimental, one would have to compensate the affected parties by a positive amount to leave them as well-off as if the event had not occurred. This is willingness to accept (WTA) compensation. Following the second point of view, the event does not occur. For a beneficial event foregone, the party would have to be compensated to make him or her as well off as if it had occurred (WTA), while if the change is harmful, the appropriate measure would be the willingness to pay to avoid it (WTP).

The first set of measures, where the event is presumed to occur, are Compensating Variations (CV). The second set are Equivalent Variations (EV). Note that either of these can represent a payment by the affected party or to the party depending on whether the change represents an improvement over the status quo or not (Table 7). Note also that WTA measures are not based by income, while WTP measures are; thus, they are not equal in general.

Table 7 - Measures of Compensation for a Change in Welfare.

	Event Happens: (CV)	Event Doesn't Happen: (EV)
Beneficial Event:	WTP To Obtain	WTA To Forego
Detrimental Event:	WTA Compensation	WTP To Avoid

In certain cases involving environmental change and conflict of interest, property rights and liability are fairly well developed. The Federal water pollution control legislation dealing with point-source discharges of known toxic substances, for example, can be taken as supportive of the property right of citizens to water of unimpaired quality. In most other cases, including nonpoint-source pollution, the implicit property rights structure is more ambiguous, and subject to periodic reinterpretation. Compensation in such cases is, at least at present, almost certain to be a hypothetical measure.

Methods for Estimating Environmental Quality Benefits

The passage of significant Federal environmental quality legislation during the 1960s and 1970s created both an interest in and a need to develop methodologies to economically value non-market commodities, particularly air and water quality improvement. Two broad categories of empirical techniques have emerged during this period: direct and indirect.

The direct methods are conceptually quite simple. Individuals are directly queried, in a standardized interview format, as to what they would be willing to pay for a specific improvement in environmental quality. The techniques are analogous to those used by private market research firms investigating the potential for new consumer goods or services. In the context of resource and environmental quality demand studies, the approach is known as the Contingent Valuation Method (CVM). The term "contingent" refers both to the hypothetical nature of the exercise (no payments are actually collected) and to the fact that studies are typically conducted ex ante, before the extent of proposed environmental quality changes are known.

The indirect methods, by contrast, rely on observable market transactions in goods with which environmental quality is associated. Indirect methods can in turn be sub-categorized as either (a) models using observations on travel behavior to impute demand curves for site-specific water-based recreation or environmental quality, known as Travel Cost models (TC); or (b) models using data on the sale or rental price of real estate, referred to as the hedonic method.

In certain instances, aspects of several different approaches are combined in one model. The hedonic travel cost method is, as the name implies, such an amalgam. In several empirical studies attempting to value the changes in water quality at a specific site, both indirect and direct approaches have been used simultaneously. These will be referred to in this document as contingent behavior models.

The Water Resources Council (WRC) has evaluated and endorsed both the Travel Cost method and the Contingent Valuation method for use in water resources project analysis. The Unit Day Value method appears as an inferior alternative. Endorsement is contingent upon proper application of the methods as well as on adequate data. When these conditions are met, the WRC concludes "...Experience indicates that the TC and CV methods can yield estimates of value with an accuracy equal to that of other project outputs", (Federal Register, 1979).

<u>Summary of Results of Empirical Studies</u>

Twenty seven empirical studies yielding estimates of the economic benefits of water quality, representing a diverse set of benefit estimation methodologies (Travel Cost, Contingent Behavior, Varying Parameter, Gravity, Discrete Choice, Contingent Valuation, Contingent Ranking, Hedonic Travel Cost, and Hedonic Property Value), were reviewed. The specific assumptions employed within the context of a given type of model, e.g., Travel Cost, were found to exhibit considerable variation. The studies were also found to represent a fairly wide range of research objectives, geographical locations, and water quality goals. Each contributes an estimate or estimates, which can be pooled and analyzed for evidence of systematic variation due to location, methodology, or other identifiable factors.

To permit comparison, the estimates emerging from the 27 studies have been standardized first, to constant-value 1986 dollars using the U.S. CPI, and second, to one or both of two units: benefits per user per day, and benefits per household (recreational party) per year. Ideally, benefit estimates obtained from Travel Cost-type models should additionally be standardized to a common assumption concerning the shadow value of the recreationists' travel time. This was not attempted due to lack of adequate primary data.

The results are summarized in tables 8, 9 and 10. The first table presents the results of studies evaluating water quality changes explicitly related to soil erosion and agricultural nonpoint-source pollution (table 8). Many of these studies were conducted in connection with the Rural Clean Water Program. The next table contains the results of several other studies judged to be of better quality, larger in scope (most are multi-site models), and representative of the state-of-the-art in water quality demand estimation (table 9). The last table summarizes estimates obtained from hedonic property value models (table 10). These models attempt to apportion the value or sale price of residential property into discrete components associated with home size, physical features, neighborhood characteristics, location, and ambient environmental quality.[5]

The value of assembling a diverse set of high quality studies is the possibility that a pattern will emerge, linking differences in empirical benefit estimates to differences in either the models and assumptions employed, location, or the specification of the environmental good (water quality change) being valued. This type of analysis, when conducted formally, is referred to as "meta-analysis" in the social sciences. "The (meta) analyses treat the results from past studies as data to "test" whether differences in the estimates (across studies) reflect systematic variations in the resources being valued or in the assumptions and methods underlying them" (Russell and Smith, 1988, p. 20).

Table 8 - Summary of Estimated Recreational Benefits From Soil Conservation Programs. (1 of 2)

Study, authors and Location	Method	Annual Visits Observed	Annual Visits for Clean Water	Benefits per Visit	Annual Benefits per Party	Total Annual Benefits	Present Discounted Value
Birch et al. Michigan (1983)	CVM	17,000	-	$1.22	$17.28 (a)	$20,750	-
Osborn & Shulstad Arkansas (1983)	TC	45,000	-	$247.40 (b)	-	$12,760,000	$89,974,000 50 Yr. 7.38%
Park & Dyer Tennessee (1985) Fishing,	TC						
Specialized:		84,900	-	$1.19	-		
General:		625,000	-	$0.30	-		
Total:		710,000	-	(c)	-	$287,145 (d)	-
Ribaudo et al. Vermont (1984)	CB/TC (e)						
Current:		30,600	56,280	$8.09	$139.55	$381,250	
Potential:			19,035	(f)	$110.05	$227,700	
Total:			75,315			$608,900	$4,107,000 (10 Yr. 7.875%)
Ribaudo et al. Vermont (1984)	CR						
Current:		-	-	$3.47	$61.27	$167,380	
Potential:		-	-	(f)	$45.38	$93,900	
Total:						$261,270	$1,763,000 (10 Yr. 7.875%)
Piper at al. S. Dakota (1987)	CB/TC						
Current:		270,900	405,000	$7.90	$106.00	$3,200,000	
Potential:			25,000		$83.75	$300,000	
Total:			430,000		(g)	$3,500,000	-

Table 8 - Summary of Estimated Recreational Benefits From Soil Conservation Programs. (2 of 2)

Study, authors and Location	Method	Annual Visits Observed	Annual Visits for Clean Water	Benefits per Visit	Annual Benefits per Party	Total Annual Benefits	Present Discounted Value
Young & Magleby Idaho (1985)	TC	-	7,575 (h)	$8.08	-	$61,080	$418,400 (50 Yr. 7.875%)
Setia & Magleby Illinois (1987)	CVM	-	-	-	$6.55	$1,584	$24,400 (50 Yr. 7.86%)
UW-IES	CVM						
Wisconsin (1986)							
Homeowners:		-	-	-	$193.00	$409,160	
Day Users:		20,740	26,420	$9.53	-	$252,000	
Potential Users:		-	-	-	$11.66	$ 42,700	
Wen	CVM						
Minnesota (1986)							
Certain Users:		-	-	-	-	$1,335,600	
Potential Users:		-	-	-	-	$150,200	
Non-Users:		-	-	-	-	$265,500	
Total:		-	-	-	-	$1,751,300	$20.9 Mill. (40 Yrs, 8%)

(a) 2,430 sample visits/172 interviews = 14.13 visits per party per year.
(b) Consumer Surplus in northern basin - CS in southern basin
(c) Unit-day values for the general recreational users were set at 1/4 specialized rates
(d) Present value of costs associated with one year's sedimentation
(e) Based on estimated 2,732 present users and 2,069 potential users. Present users make 11.2 annual visits to bay in current condition, 20.6 if clean. Potential users make 9.2 visits if clean.
(f) Total annual benefits/total annual visits if clean, all users
(g) Based on results of Vermont study: potential users benefits average 79% of current users
(h) Additional user days for cleaner water. No figures for current use available
Entire CS for additional user-days attributed to water quality improvement

CVM = Contingent Valuation Method
TC = Travel Cost
CB = Contingent Behavior
CR = Contingent Ranking

Table 9 - Summary of Estimated Benefits From Water Quality Improvement.

Study authors, date and location	Method	Water Quality Change	Number of Sites	Benefits per Visit	Annual Benefits per User	Annual Benefits per Site	Aggregate Annual Benefits	Comments
				1986 U.S. $				
Bouwes & Schneider Wisconsin (1979)	TC	LCI 10 to LCI 3	1 (a) (8)	$ 2.75	$ 38.00	$ 75,050	$ 75,050	$ 640,000 (20 Years @ 10%)
Feenberg & Mills Boston (1980)	Discrete Choice	10% Reduction in Oil, Discoloration, and Bacteria	29	$ .15 (b)	$ 1.55	$ 141,000 (c)	$ 4,090,000	
Feenberg & Mills Midwest (1980)	Aggregate Tobit	"Dirty" to "Fair" "Fair" to "Clean" "Dirty" to "Clean"	292 " "	$ 3.39 $ 1.79 $ 5.18	- - -	$ 3,900,000 $ 2,050,000 $ 5,950,000 (d)		
Sutherland Pacific NW (1982)	Gravity / TC	Fishable to Swimmable	179	-	-	-	$ 4,300,000 $ 3,900,000 $ 7,800,000 $ 5,300,000	(Swimming) (Camping) (Fishing) (Boating)
Russell & Vaughan U.S. (1982)	Participation / TC	BPT (g) BPT+Erosion Control BAT	U.S.	-	$ 14.00 ($3.90) $ 16.20 ($4.50) $ 19.80 ($5.50) (e,f)	- - -	$ 830 Million $ 960 Million $ 1175 Million (f)	
Smith & Desvousges Penn. (1983)	Varying Parameter I	Boatable to Fishable Boatable to Swimmable	13 (h) "	$ 1.14 $ 2.36	$ 8.17 $ 17.15	- -	- -	
" " U.S. (1983)	" "	Boatable to Fishable	21	$ 33.62 (j)	$ 165.50	-	-	
Smith & Desvousges U.S. (1985)	Varying Parameter II	Boatable to Fishable	21	$ 3.78 (i)	$ 20.28	-	-	
Smith & Desvousges U.S. (1986)	Generalized TC	Boatable to Fishable Boatable to Swimmable Preserve Boatable	13 " "	- - -	$ 8.63 $ 34.80 $ 4.25	- - -		

Table 9 - Summary of Estimated Benefits From Water Quality Improvement.

Study authors, date and location	Method	Water Quality Change	Number of Sites	Benefits per Visit	Annual Benefits per User	Annual Benefits per Site	Aggregate Annual Benefits	Comments
				1986 U.S. $				
Smith & Desvousges Penn. (1987)	CVM	Boatable to Fishable Boatable to Swimmable Preserve Boatable	13 " "	- - -	$ 22.50 $ 41.60 $ 24.00 (j,k)	- - -	- - -	
Carson & Mitchell U.S. (1988)	CVM	Preserve Boatable Boatable to Fishable Fishable to Swimmable	U.S. " "	- - -	$ 102.00 $ 76.90 $ 85.70	- - -	$ 22.3 Billion (k)	
Graham-Tomasi Michigan (1986)	Discrete Choice (ML)	Damages from 1-time discharge of sediment	2 (Counties)	-	-	-	$285,000.00	3-Year Period
Huang Minnesota (1986)	Varying Parameter	1-Point Improvement in Weighted Index	22	$1.01 - $5.93 (Avg $2.40)	-	-	-	
Huang Minnesota (1986)	Hedonic TC	1-Point Improvement in Weighted Index	22	$1.35 - $7.21 (Avg $3.19)	-	-	-	
Graham-Tomasi et al. Minnesota (1986)	Discrete Choice (ML)	Loss of .8% - 1.4% of Statewide Fishable Acreage	Minn.	$0.27	-	-	$1.6 Million	

(a) Demand equation was estimated over 8 sites, benefits calculated for 1.
(b) Based on observed average 10.3 visits per capita per year.
(c) Aggregate benefits distributed equally over 29 sites.
(d) Per-visit benefits multiplied by catchment population.
(e) Annual benefits per angler (annual benefits per capita.)
(f) Annual party and annual aggregate benefits represent average of high and low valuation bases.
(g) As specified by FWPCA/CWA.
(h) 13 sites on Monongahela River: parameters estimated for 21 Corps of Engineers sites.
(i) Benefits per trip, not necessarily 1 day in duration.
(j) Sample-weighted average of 4 CVM formats.
(k) Both per capita and national aggregate benefit estimates include non-user values, including option, preservation, and bequest values.

ML = Maximum likelihood
CVM = Contingent Valuation method
TC = Travel Cost

Table 10 - Summary of Estimated Benefits, Hedonic Property Value Studies.

Study Authors and Location	Sample Size	Specified Water Quality Change	Average Property Value Change	Change as % of Average Property Value	Regional Benefits
			1986 U.S. $		
Epp & Al-Ani Penn. (1979)	212	Continuous	$ 1,960	5.95 %	-
Feenberg & Mills Boston (1980)	506 (a)	10% Reduction in Oil and Turbidity	$ 6.71 (b)	-	$ 7.2 Million
Rich and Moffit Mass. (1982)	42	Observed Improvement 1968-1975	$ 44.30/acre (d)	-	$ 720,000
			$ 37.10/acre (e)	-	
Young & Teti Vermont (1984)	93	Bay Quality Equal to Lake Quality	$ 6,025 (f)	21.0 %	$ 2,590,000
			$ 5,385 (g)	18.75 %	$ 2,315,000

(a) Census tracts.
(b) This figure is expected to differ form other results since the Boston study did not attempt to value the setting of properties, but rather estimated the amenity benefits of improved water quality at recreational beaches as capitalized in regional home values.
(c) Based on 2.63 million 1970 Boston residents over 18 years old, and 2.46 such persons per household on average.
(d) Riparian
(e) Non-riparian
(f) Estimated using location indicator for water quality.
(g) Estimated using continuous water quality variable.

Smith and Kaoru (1988) conducted a formal meta-analysis of this type to examine the determinants of estimated recreational resource values. They reviewed over 200 Travel Cost demand models prepared between 1970 and 1986, which yielded 734 individual observation on Consumer Surplus (CS) per unit of use. The estimates refer to the use-value of a recreational resource, and not the value of environmental quality change, but the methodology used and results obtained are instructive for this investigation. Smith and Kaoru's model took the form:

$$RCS_i = \alpha_s X_{si} + \alpha_a X_{ai} + \alpha_d X_{di} + \alpha_e X_{ei} + \epsilon^i$$

where RCS_i is the real (constant value) consumer surplus per unit of use of a recreational site by individual i, X_{si} a vector of characteristics of the sites visited by i, X_{ai} the assumptions inherent in the model (shadow value of travel time), X_{di} the functional form of the demand model (linear, semi-log, etc.), and X_{ei} the type of estimator (OLS, GLS, ML). Given the wide variety of characteristics of sites and studies, transferability of a "representative" benefit estimate generated at one location to a second location would require that $\alpha_a = \alpha_d = \alpha_e = 0$. Restated, "...judgmental modeling assumptions contribute to the variability in benefit estimates but do not impose systematic influences on the size of the benefits estimated" (Russell and Smith, 1988, p. 24).

Smith and Kaoru's data and conclusions are summarized in Tables 11 and 12. The first table gives evidence of extreme variability in estimates of CS per unit of use. In four of the seven categories, estimates range over two orders of magnitude, and in one case over three

Table 11 - Comparison of Travel Cost Demand Results by Type of Resource.

Type of Resource	Number of Estimates	Real Consumer Surplus[a] Mean	Range
River	257	$17.05	$0.29 - $120.70
Lake	483	$16.85	$0.09 - $219.80
Forest	114	$31.36	$0.80 - $129.90
National Parks	12	$44.01	$23.48 - $120.70
Wetlands	9	$45.86	$17.45 - $120.70
State Parks	107	$42.49	$0.67 - $327.20
Coastal Areas	28	$35.49	$0.67 - $160.80

(a) Real consumer surplus deflates the nominal estimates by the consumer price index, base 1967.

Source: C. S. Russell and V. K. Smith (1988), Table 8 page 53.

Table 12 - Determinants of Real Consumer Surplus Per Unit of Use.

Independent Variables	Models: # 1	# 2	# 3	# 4	# 5
Intercept	23.72	16.07	20.30	27.03	18.75
	(5.62)	(2.08)	(6.19)	(3.68)	(0.58)
Surtype	7.99	-4.13	-9.97	15.38	19.88
	(2.76)	(-1.45)	(-2.72)	(2.97)	(3.74)
<u>Type of Site (X_s):</u>					
Lake	-11.70	-	-	-18.69	-20.32
	(-3.18)	-	-	(-3.24)	(-3.52)
River	-5.57	-	-	-14.29	-19.03
	(-1.93)	-	-	(-2.99)	(-2.19)
Forest	-0.45	-	-	-18.45	-25.99
	(-0.93)	-	-	(-2.36)	(-3.01)
State Park	19.93	-	-	24.95	22.37
	(4.44)	-	-	(3.47)	(3.44)
National Park	2.54	-	-	0.56	-3.77
	(0.20)	-	-	(0.04)	(-0.23)
<u>Model Assumption (X_a):</u>					
Substitute Price	-	-	-18.73	-	-13.71
	-	-	(-3.27)	-	(-2.12)
Opportunity Cost Type #1	-	-	-14.97	-	-16.49
	-	-	(-2.10)	-	(-2.11)
Opportunity Cost Type #2	-	-	3.95	-	-15.86
	-	-	(1.20)	-	(-3.30)
Fraction of Wage	-	-	37.24	-	48.59
	-	-	(8.56)	-	(9.76)
Specific Site/ Regional TC Model	-	-	22.23	-	24.21
	-	-	(4.10)	-	(3.85)
<u>Model Specification (X_d):</u>					
Linear	-	2.35	-	-	-2.87
	-	(0.31)	-	-	(-0.27)
Log-Linear	-	14.63	-	-	23.37
	-	(1.89)	-	-	(2.37)
Semi-Log	-	11.26	-	-	16.89
	-	(1.52)	-	-	(1.86)

Table 12. - Determinants of Real Consumer Surplus Per Unit of Use.

Independent Variables	Models: # 1	# 2	# 3	# 4	# 5
Estimator (X_e):					
OLS	-	-	-	-	-14.65
	-	-	-	-	(-0.48)
GLS	-	-	-	-	-8.58
	-	-	-	-	(-0.28)
ML-Trunc	-	-	-	-	-67.38
	-	-	-	-	(-2.15)
R^2	.11	.03	.25	.15	.42
n	722	722	399	399	399

Source: Russell and Smith (1988), Table 9, pp. 54-56.

orders of magnitude. This extreme variability is also observed between the 27 water quality benefit studies examined above. While it is possible that differences in CS are solely attributable to differences in site characteristics, regression model results suggest otherwise (Table 12). Specifically, it is seen first that equation 1, containing site-related characteristics (X_{si}) exclusively as predictors, explains very little of the variation in CS estimates (R^2 = .11.) The preferred model (equation 5), which provides the greatest explanatory power, presents strong evidence that model assumptions (X_{ai}), and to a lesser extent specification (X_{di}) and choice of estimator (X_{ei}) act to determine the level of estimated CS. Note particularly how influential the assumption is concerning the shadow value of the recreationists' travel and on-site time (fraction of wage).

The 27 studies reviewed in this chapter embody two additional sources of variation not analyzed by Smith and Kaoru. First, at least three distinct methodologies have been employed: Travel Cost, Contingent Valuation, and Hedonic Property Value (Smith and Kaoru examined only Travel Cost models). The second concerns the various assumptions on base level and magnitude of change in water quality. This distinction is non-trivial in that the non-market good being valued is a specific change in water quality at a specific site, and not recreational use of the site per se. This raises a considerable obstacle to meta-analysis, even on a level less formal than Smith and Kaoru's. In several cases (Ribaudo et al., 1984, Piper et al., 1987, Young & Magleby, 1985), no objective measure of either baseline water quality or proposed water quality change was available. Over the remaining studies, at least six independent water quality measures were employed, and with varying degrees of precision and quality measures were employed, and with varying degrees of precision and consistency. This, and the absence of individual recreator data for many of the studies reviewed, precludes the estimation of a model analogous to Smith and Kaoru's. It also makes summary statistics on benefit estimates (mean, standard deviation) misleading or irrelevant.

Some insight can be gained, however, by arraying the standardized benefit estimates in "dollar - water quality space." Benefits per user (party) per day (visit) are illustrated in Figure 3, and benefits per recreator party per year or season appear in Figure 4. The key to the vertical water quality scale is shown in Figure 5. The vertical bars associated with each study, identified by both geographical region and basic study methodology, are located so that the lower end corresponds to the baseline or pre-project water quality level, and the upper end to the proposed or post-project water quality level. Bars ending in horizontal lines identify studies summarized in Table 8; those ending with circles identify the more comprehensive studies summarized in Table 9. The length of the bar thus provides an indication of the magnitude of the water quality change. Studies not yielding benefits in user-day or annual party terms (Wen, 1986, Setia & Magleby, 1987), studies basing benefits on changes in the _quantity_ of water of a given quality (Graham-Tomasi, 1986, Graham-Tomasi, et al., 1986), and obvious outliers (Osborn & Shulstad, 1983), are not represented in Figures 3 and 4.

The procedure used to put the various water quality indicators on a common scale is _ad hoc_ and it must be recognized that there is no means to establish a scientifically valid, strictly one-dimensional correspondence between the indicators. Objective water quality indicators are often

Figure 3 – Summary: Benefits per Party per Day From Water Quality Improvement

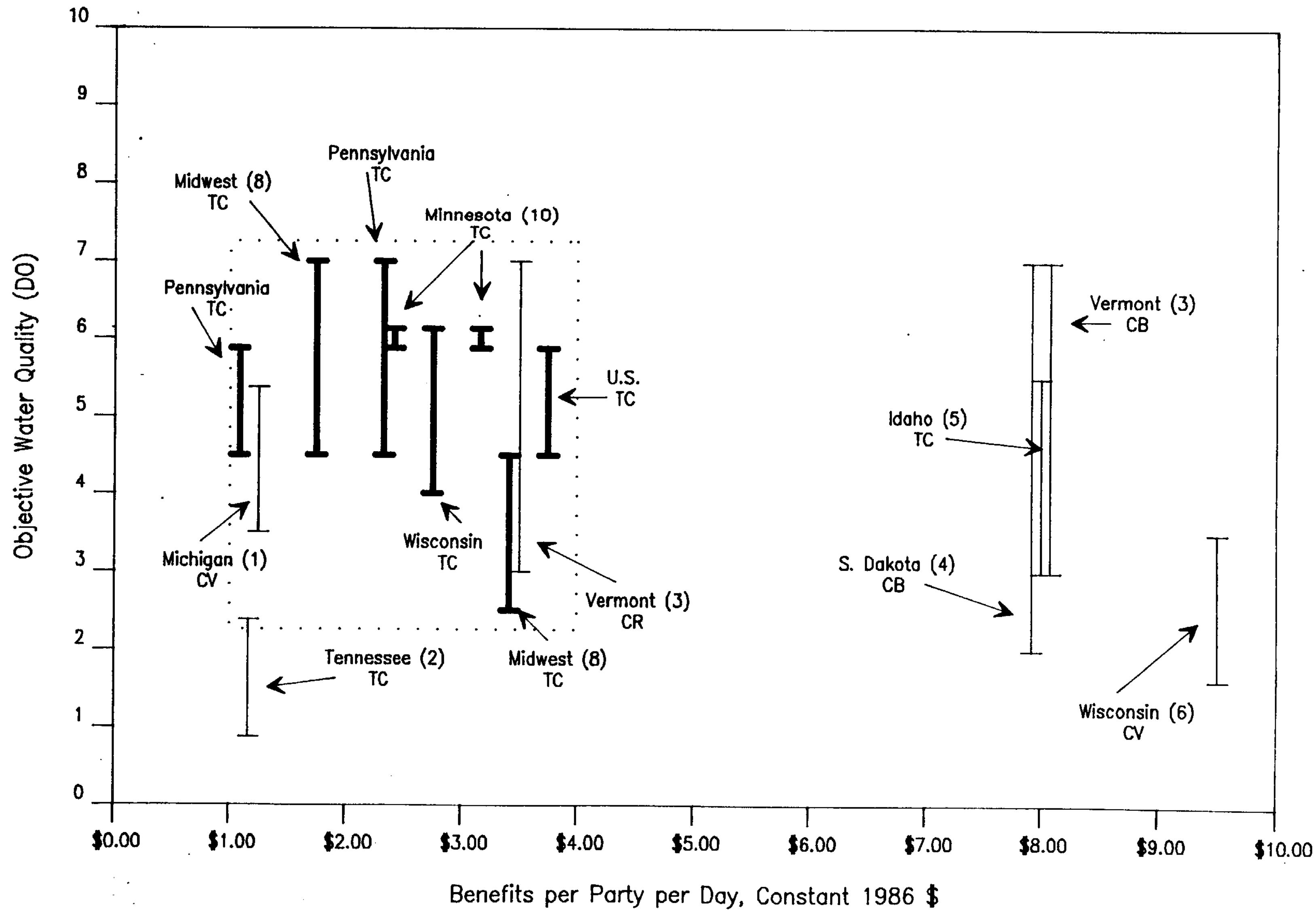


Figure 4 – Summary: Benefits per Party per Year From Water Quality Improvement

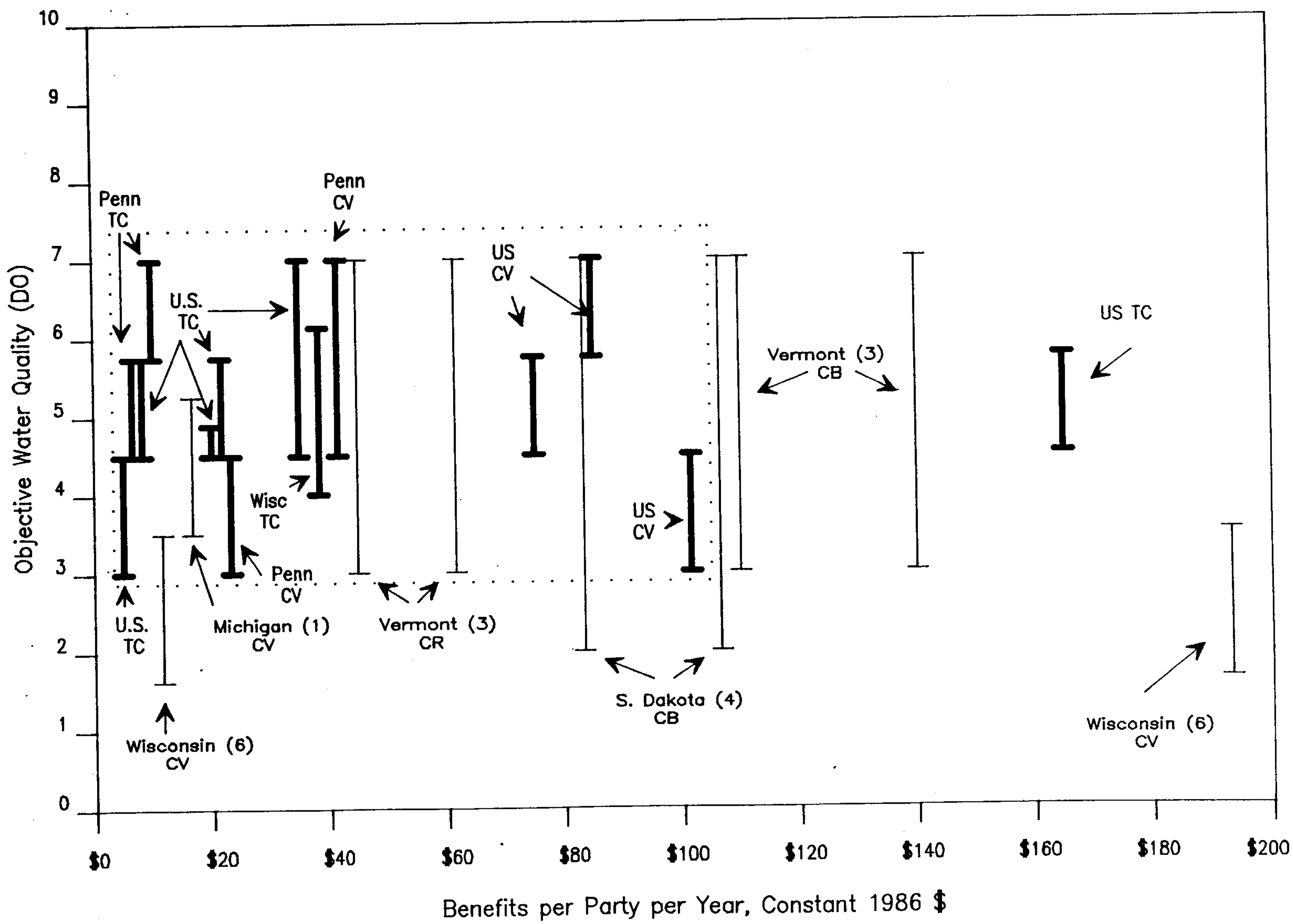
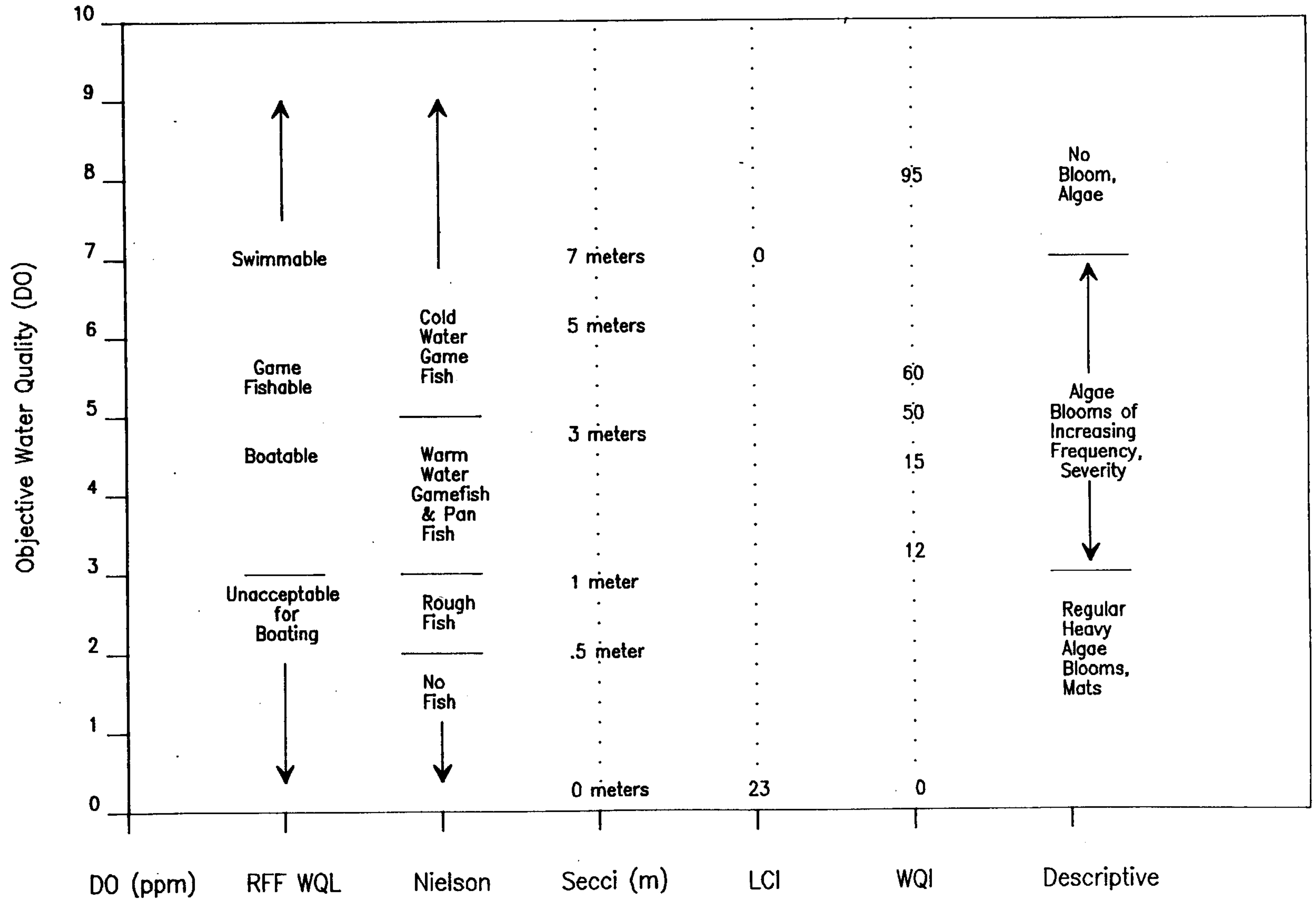


Figure 5 – Key to Water Quality Measures



devised to evaluate the suitability of water for specific uses, such as fishing or drinking. They typically embody several independent attributes, including Dissolved Oxygen (DO), nutrient (N,P) content, temperature, pH, and turbidity. Dissolved Oxygen, in ppm (equivalently in mg/litre) was selected as the single best quantitative indicator for comparison between studies, and is used to index the 1 to 10 Water Quality scale on the Y axis of Figures 3, 4 and 5. Dissolved Oxygen is an important component of nearly all established water quality indices (e.g., LCI, WQI) and is essential for fish species survival and dominance thresholds. The RFF Water Quality Ladder, used in many of the studies, is keyed to DO.[6]

If, overall, benefits were proportional to the magnitude of the water quality change, the longer bars would generally be arrayed to the right of the shorter bars in Figures 3 and 4. Similarly, if water quality changes at the higher end of the water quality spectrum (e.g., game fishable to swimmable) were valued more highly (less) than quality changes lower on the spectrum, one would expect a pattern of bars sloping upward (downward) to the right. Neither pattern is discernable in Figures 3 and 4. The distribution of benefits also appears essentially random with respect to geographical location.

One apparently influential factor is the type of demand estimation procedure used: indirect (TC) vs. direct (CVM,CB,CR).[7] Contingent Valuation studies appear to yield systematically higher benefit estimates than TC studies. This is not unexpected, since CVM studies can capture non-use values, while TC models cannot.

It is also evident that the single-site models focused on agricultural nonpoint-source pollution as the source of water quality impairment yield higher and more varied benefit estimates than the more generalized multi-site models (Tables 8 and 9). Taken together, benefits per user-day range from $1.00 to $10.00, and per recreational party annually between $3.00 and $200.00, with more observations at the lower end of the range. The multi-site models (circles at end of bars) describe more narrow intervals of benefit estimates: $1.00 to $4.00 per user day and $4.00 to $100.00 annually per party. For reasons relating to both sample size and methodology, these more conservative estimates are probably more reliable.

Non-use Values

The majority of recreational demand studies summarized in Tables 8, 9 and 10 have as their objective the estimation of benefits to users and potential users of recreational water resources associated with maintenance or improvement in water quality. That is, benefits are the change in consumer surplus associated with water <u>use</u> and its corresponding change in quality. A subset of these studies, the CVM models (Carson and Mitchell, 1988; Wen, 1986; Desvousges, Smith and Fisher, 1987) attempted to measure benefits that are not directly associated with recreational use of water resources. These benefits are associated with the values of preserving the option to utilize the resource in the future (option and quasi-option values), knowing the resource is available for others to use (vicariousness), insuring that those resources pass intact to heirs or future generations (bequest), and knowing that the integrity of the aquatic ecosystem is being protected or improved, for the benefit of the ecosystem per se (existence, stewardship values).

Few resource economists currently challenge the assertion that such non-use benefits exist. Substantial controversy remains, however, over the precise definition and empirical measurement of many non-use values. To include such values in the benefit-cost analysis of water-related projects requires, minimally, that (a) proposed benefits meet the normative criteria of benefit-cost analysis, i.e., that they arise from individual preferences in a manner consistent with economic theory; (b) they represent distinct, non-overlapping categories of value, and (c) they are empirically measurable.

An additional obstacle to obtaining accurate estimates of non-use values is the inability of indirect or travel cost-based models of recreational behavior to measure them. Measurements can be obtained from direct (CVM) studies, but are subject to multiple sources of bias. A number of researchers (e.g., Smith et al., 1986, Ribaudo et al., 1984, Wen, 1986), have estimated both TC and CVM models for a common sample of respondents, so that the results can be cross-validated, an option not available where non-use values are concerned. Distinctions between categories of non-use value are also often elusive. In addition, the contingent valuation responses may be extremely sensitive to the wording of questions and to the ability of respondents to conceptually distinguish between categories of value.

The existence of unresolved issues associated with the definition and measurement of non-use values should not be interpreted as evidence that benefit-cost analysis should be restricted to use-related benefits. Accumulating evidence from CVM studies suggests that this will result in an under-investment in projects designed to protect and enhance environmental quality. The variation in these estimates, and the obvious methodological flaws in many studies should, however, alert project managers to proceed with caution whenever projected non-use benefits appear to be a large percentage of total projected benefits, or when their exclusion causes rejection of a project that would be accepted otherwise. The results of several empirical studies representing a wide range of research settings are summarized in Table 13 (Fisher and Raucher, 1984).

Off-Site Non-Recreational Benefits of Conservation

Recreational and non-user benefits from agricultural NPS abatement have been described, and quantified, as large and significant components of total expected off-site benefits. A more complete taxonomy includes several other benefit (damage) categories encompassing in-stream, storage and conveyance, and withdrawal uses of surface water, as well as benefits produced by general ecosystem stability and diversity.

Benefit estimation methodology has been developed for several of the subcategories, and empirical estimates obtained. These subcategories include water storage and conveyance, flood damages, irrigation and agricultural water management, and water treatment. Most of the methodologies are not demand-based, but rather represent the "engineering approach with observed averting behavior" (Courant and Porter, 1981). The engineering approach to valuation differs from demand-based valuation techniques in that it focuses exclusively on physical or technological relationships, and ignores individual behavior, preferences, or measures of (unobserved) surplus. Market prices are used rather than WTP or similar measures in determining costs, consequently the engineering approach is a lower bound estimate of costs (benefits) associated with

Table 13 - Empirical Estimates of Non-Use Values.

Study, author and date	Site:	Estimation Technique:	Estimates: $ per Household per Year $	Use ($)	Non-Use ($)	Ratio Nonuse/Use	Comments:
Meyer (1974)	Fraser River, British Columbia	Residents surveyed regarding recreation and preservation values of salmon	1981	928	502	0.54	-use values include aesthetic benefits for near-stream activities. -option value omitted.
Hovarth (1974)	Southeastern U.S.	Residents surveyed as to value of wildlife attributes	1981	2824	1574	0.56	-nonusers defined as "those who would have liked to but for some reason did not participate in the study year. -values presented are averages for fishing.
Dornbusch and Falcke (1974)	7 locations in U.S.	Property owners surveyed as to relative importance of water quality improvement attributes	1981	-	-	.75-2.03	-lower value ratio based on inclusion of aesthetic values in use benefits. -higher value ratio based on inclusion of aesthetic values in nonuse benefits.
Meyer (1978)	Fraser River British Columbia	Update of 1974 study with improved sampling and survey instrument	1981	287	360	1.26	-same as Meyer (1974).
Walsh et al. (1978)	S. Platte River Colorado	Residents surveyed via bidding game on value of improved water quality	1981	126	66	.53	-use value is option price. -nonuse value is the sum of existence and bequest value to those not expecting to use the basin for recreation.
Mitchell & Carlson (1981)	U.S. (Nationwide)	National survey of recreators and nonusers of surface waters	1981	258	121	.47	-values presented are for "fishable" quality. -use value equals total bid of recreators (use + user intrinsic values). -nonuse defined as nonparticipation in in-stream activities within past two years.

Table 13 - Empirical Estimates of Non-Use Values.

Study, author and date	Site:	Estimation Technique:	$	Use ($)	Non-Use ($)	Ratio Nonuse/Use	Comments:
			Estimates: $ per Household per Year				
Desvousges, Smith & McGivney (1983)	Monongahela River	Residents surveyed via bidding game, direct question, and question with payment card on value of water quality	1981	52	34	.65	-prevent loss of boatable quality.
				62	28	.44	-improvement to swimmable quality.
Cronin (1982)	Patomac River	residents surveyed via direct question for value of water quality	1982	42	30	.71	-prevent loss of boatable quality.
				125	84	.67	-improvement to swimmable quality.
Cronin (no date)	Patomac River	Same as above	-	44	35	.80	-as above, but nonusers defined as those who would not use the Patomac even if it were as clean as they would like it to be. Users are present users and those who would use a cleaned-up river.
				137	66	.48	
Walsh, Loomis, & Gillman (1984)	Colorado	Residents surveyed by mail regarding wilderness preservation	1984	19.44	13.31	.68	-use includes option value. -nonuse includes existence and bequest.
Sutherland & Walsh (1985)	Flathead Lake, Montana	Residents surveyed by mail regarding water quality preservation in lake and rivers	1985	18.08	46.25	2.56	Same as above.

Source: Fisher and Raucher (1984), p.50-51 and 59; Walsh, Loomis and Gillman (1984) p.25; Smith and Desvousges, p.130.

sedimentation (abatement).

Wen (1986) used the engineering approach and a sediment-routing model to estimate damage costs per acre of cropland associated with sediment removal in the Lower-Upper Mississippi river (table 14). The sediment delivery ratio (SDR) refers to the percent of gross erosion in each sub-basin that reaches the lower-upper Mississippi channel. Wen distinguishes between total sediment delivery, and loads associated specifically with human activities.

Several researchers have estimated the costs of removing eroded sediment from roads, drainage ditches, and culverts (see table 15). A relatively high percentage of dislocated sediment is re-deposited in edge-of-field ditches, which must be cleaned periodically. Damage costs are taken as equal to the costs of removal, but actual damages are probably higher, since budget constraints rather than extent of need often determine the frequency and extent of ditch and roadway clearing.

Incremental flood damages can also be attributed to cropland erosion, resulting from (i) greater sediment load of floodwaters, (ii) sediment contribution to higher floodwater volume, and (iii) sediment-caused channel aggradation, leading to increased frequency of flooding. Clark II, et al. (1985), summarizes estimated nationwide flood-related damages from soil erosion (table 16).

Off-site damages of all types are expected to exhibit considerable variation with regard to location, and not all proposed categories of offsite damages (benefits) will be applicable in a given watershed or region. A range of combined damages (benefits) per acre from soil erosion (conservation) has been estimated by Ribaudo (1986b). Ribaudo

Table 14 - Net Dredging Costs from Soil Erosion in the Lower-Upper Mississippi.

	Sub-Basins			
	Cannon River	Zumbro River	Whitewater River	Root River
SDR (%)	3.45	2.17	17.82	3.35
Net Dredging Cost ($)	23,500	78,470	12,185	586,090
Total Cropland (acres)	559,180	641,200	55,526	676,454
Avg. Per Acre Damage ($)	.042	.122	.219	.866
Man-Induced Erosion (%)	92.4	88.02	87.32	78.46
Man-Induced Damage per acre ($)	.039	.108	.192	.680

Source: Modified from Wen (1986), Tables VII-4 p. 309, VII-13 p. 339.

Table 15 - Cost Estimates for Removal of Sediment from Roads, Ditches, and Culverts.

Study author and date	State	Annual Statewide Damages or Costs	Costs per Acre	% of Gross Erosion
Taylor et al. (1972)	Illinois	$ 8.0 million (1980 $)	-	1.5%
Lee et al. (1974)	Illinois	-	varies	22.5 - 37.5%
Forster & Abrahim (1985)	Ohio	$ 1.0 million (1985 $)	$ 0.45	8.4%
Michalson, Brooks (1984)	Idaho	-	$ 2.35	3.0%
Fletcher (1986)	Indiana	$ 9.5 million (1986 $)	-	-
Moore & McCarl (1987)	Oregon	$ 4.2 million (1987 $; for Willamette Valley)	$ 2.00	-

Table 16 - Sediment-Related Flood Damages, Millions of 1980 $.

Type of Impact	Range of Estimates	Point Estimate	Cropland's Share
Increased Flood Heights from Channel Aggradation	$ 0 - $ 190	$ 50	$ 16
Increased Volume	10 - 50	23	8
Direct Sediment Damage:			
Urban	260 - 510	350	110
Other	160 - 330	250	82
Loss of Life	14 - 33	-	-
Reduced Agricultural Productivity	0 - 170	100	33
Total (rounded)	440 - 1300	770	250

Source: Clark et al. (1985) p.166 Figure 5.12

disaggregated the summary estimate of national damages proposed by Clark et al. (1985) to the Farm Production Region (FPR) level using a variety of weighting techniques.

Ribaudo's (1986b) estimates, adjusted to 1986 dollars, are summarized in Table 17. Combined off-site damages range from a low of $0.54 per ton of soil eroded in the Northern Plains to $6.11 in the Northeast, with a national average of $1.52. Benefits from 1983 conservation practices range from $0.31 per ton of soil (Northern Plains) to $1.64 (Northeast) with a nationwide average of $0.61. Damages (benefits) per ton of soil eroded (conserved) appear inversely related to gross erosion at the level of FPR's. The FPR's with the highest damages, and potential benefits per ton of soil are those with higher population densities (Northeast, Delta States, Lake States, Pacific) and correspondingly higher water demand for both instream and withdrawal uses. These results support the assertion that off-site economic damages (benefits) from soil conservation programs primarily reflect the demand for water and water quality rather than the level of gross erosion.

MANAGEMENT PRACTICES AND SEDIMENT/NUTRIENT DELIVERY

Having identified a range of values within which benefits from water quality improvement are likely to be found, the next task is to attempt to answer questions concerning the delivery of this environmental good. Off-site economic benefits from soil conservation will be obtained only if conservation practices are effective in influencing surface water quality. Owing to the sheer volume and complexity of the issues involved, the following review is highly selective.

Table 17 - Off-Site Erosion Damages and Benefits From 1983 Soil Conservation Programs By Farm Production Region.

Farm Production Region	Erosion, All Sources 1,000 Tons	Damages, All Sources 1,000 $	Damages Per Ton $	Erosion Reduction 1,000 Tons	Off-Site Benefits 1,000 $	Benefits Per Ton $
	- - - - - - -	- - - - - - -	- 1986 $ -	- - - - - - -	- - - - - - -	- - - -
Appalachian	484,000	582,000	$1.20	66,000	38,900	$0.59
Corn Belt	970,000	1,019,100	$1.05	94,100	29,700	$0.32
Delta States	234,000	531,500	$2.27	26,800	32,000	$1.19
Lake States	181,000	568,900	$3.14	11,100	16,900	$1.52
Mountain States	1,003,000	892,800	$0.89	72,500	35,800	$0.49
Northeast	185,000	1,131,100	$6.11	7,800	12,800	$1.64
Northern Plains	671,000	361,300	$0.54	64,900	20,300	$0.31
Pacific	669,000	1,482,500	$2.22	30,600	26,000	$0.85
Southeast	250,000	352,500	$1.41	47,700	41,200	$0.86
Southern Plains	490,000	861,000	$1.76	189,600	119,400	$0.63
Total:	5,137,000	7,782,700	$1.52	611,100	373,000	$0.61

Source: M. Ribaudo, (1986b), Tables 10, 15.

There are numerous variables and pathways influencing Non-Point Sources (NPS) pollutant delivery and the erosion rate is but one of the factors determining downstream pollutant loadings (see Figure 6). It follows that "...the ranking of individual fields for erosion potential is not necessarily the same as the ranking for potential surface runoff (and loading contribution) given the same precipitation events" (Robillard and Walter, 1983, p. 332).

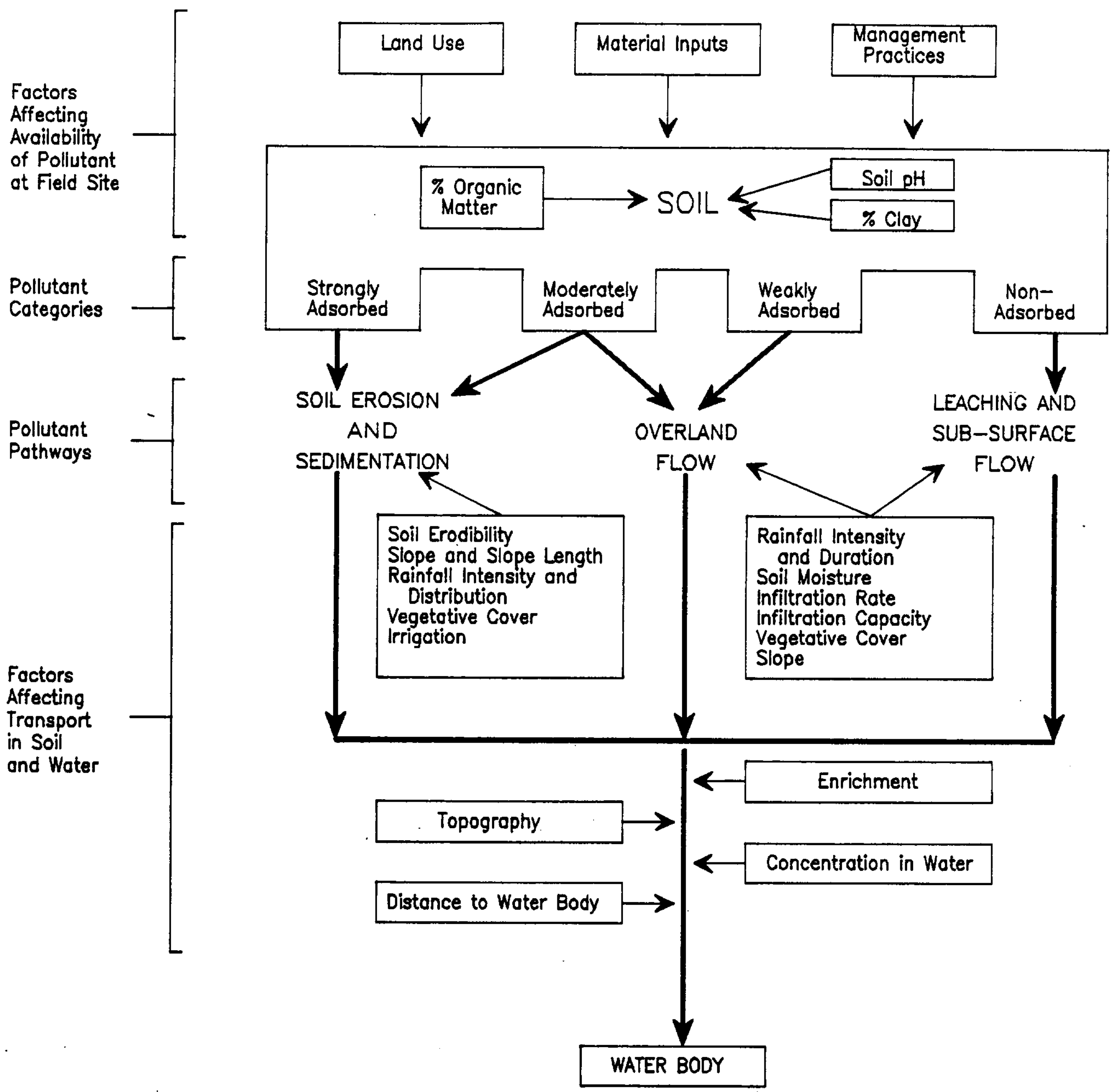
In addition, all land uses and management practices, including permanent cover, are associated with some positive loadings of sediment and nutrients. Forest land, for example, contributes significant quantities of BOD and other nutrients from decaying organic matter. The presence of this background or base load insures that no set of BMP's, including the removal of land from cultivation, will totally eliminate sediment and nutrient delivery. Total Pollution Flux (TPF), or gross changes in pollutant delivery due to human activity, may be small relative to overall loading levels in some watersheds.

The most basic loading function for sediment from sheet and rill erosion is the well-known USLE (Wischmeier and Smith, 1978), combined with estimates of the sediment delivery ratio (SDR) (EPA, 1976a):

$$Y(S)_E = \Sigma_i[A_i(R \cdot K \cdot L \cdot S \cdot C \cdot P \cdot SDR)_i]$$

Here, $Y(S)_E$ is sediment loading in mass per unit time, A_i is source area i (the loading function sums deliveries from i=1..n homogenous sub-units), R,K,L,S,C,P are the rainfall, soil erodability, slope length, slope gradient, cover, and practice factors of the USLE, and SDR the sediment delivery ratio. The USLE by itself can provide reasonably

Figure 6 – Variables and Pathways Influencing Pollutant Delivery



Source: Robillard and Walter (1983), Figure 17.2, pp. 333.

accurate estimates of soil dislocation or sediment production on unit areas of relatively homogenous characteristics and for a rainfall pattern of known intensity. However, sediment produced or dislocated will be re-deposited if production exceeds the sediment transport capacity of overland flow. Thus the quantity of sediment delivered to the watershed or sub-basin outlet will necessarily be less than the total produced ($0 \leq SDR \leq 1$).

The SDR, typically calculated at the watershed or sub-basin level, will be determined by the watershed size, drainage density (stream miles/area drained in $miles^2$), soil characteristics (e.g., size and density distribution of particles,) topography, energy and intensity of weather events, and to a lesser extent, land use. The interaction of SDR, drainage density and soil type is captured in Figure 7.

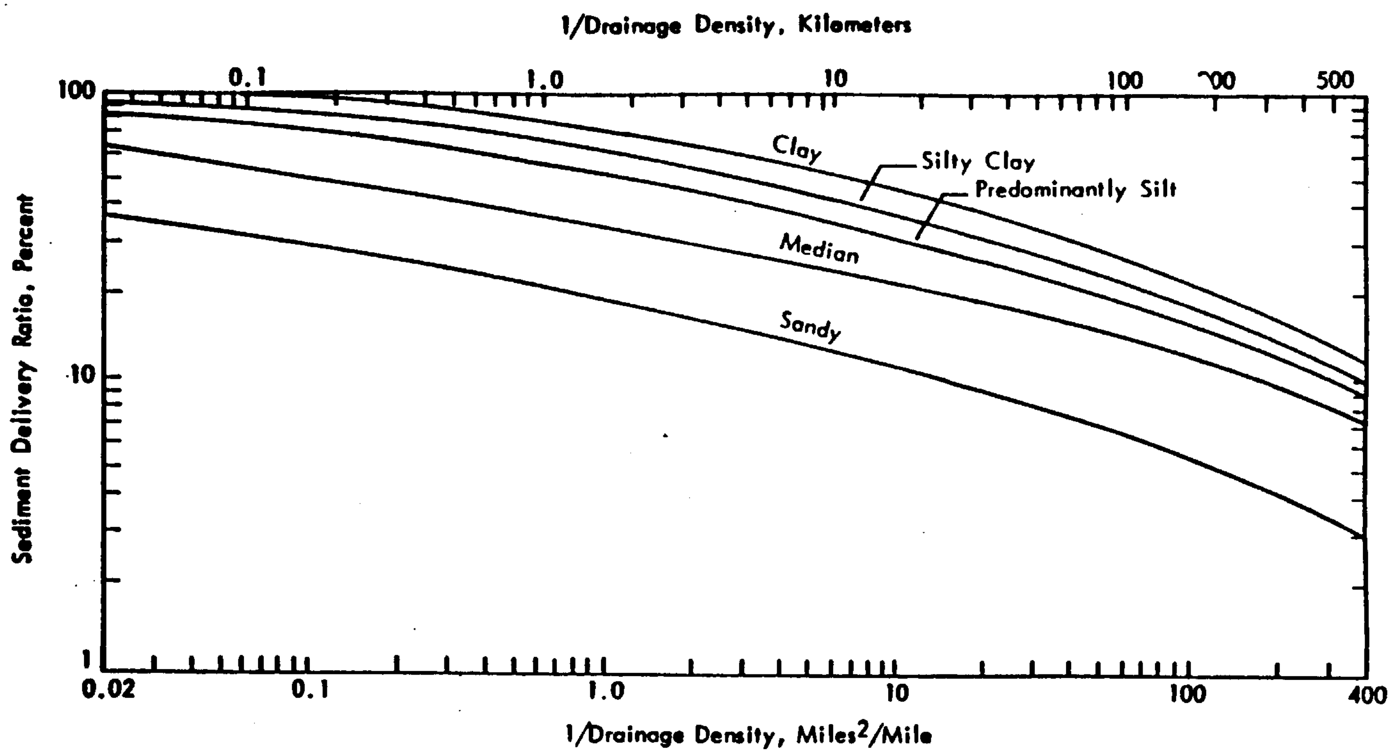
There is some variation in the SDR introduced by land use, which will influence infiltration and sediment transport capacity, among other factors. Resources for the Future has estimated statewide average SDRs for several classes of land use (see Table 18) (Gianessi, et al., 1985).

The estimated values and their distribution suggests that shifts in land use, and by implication management, have little effect on the mean SDR. This suggests that agricultural BMP's will lead to reductions in downstream loadings of sediment and sediment-associated pollutants primarily through their effect on gross erosion.

Other approaches have been suggested. Williams (1972) modified the USLE to permit calculation of sediment yield as a function of storm intensity factors. His modified USLE (MUSLE) is estimated as:

$$A = 95(Qq_p)^{0.56}(K\ L\ S\ C\ P)$$

Figure 7 - Sediment Delivery Ratio for Relatively Homogenous Basins



Source: EPA (1976) Loading Functions for Assessment of Water Pollution from Nonpoint Sources, Figure 3.10, p. 66.

Table 18 - Estimated Sediment Delivery Ratios by Land Use Category.

Land Use:	n. obs.	Range of Estimates	Mean	Std Dev
Cropland	48	.22 - .58	.452	.082
Pasture	48	.18 - .61	.441	.083
Rangeland	22	.17 - .60	.429	.099
Forest	48	.20 - .58	.403	.078
Other Land	48	.20 - .58	.404	.100

Source: Gianessi et al. (1985), Table 9, p. 17.

where A is sediment yield from a given unit area and for a given storm event, Q is total storm runoff volume in acre-feet, and q_p is peak runoff rate in feet^3/sec. All the other variables (KLSCP) are specified as in the USLE. The modified form captures the positive relationship between the SDR and increased storm intensity, through increased overland sediment transport capacity. Sediment dislocated by storms with low intensity and/or low overall runoff is more likely to be re-deposited before reaching waterways.

Another alternative approach to estimating sediment loading is the use of parametric or deterministic models. These models focus on detachment, transport, and deposition as discrete processes, and employ numerical coefficient estimates for each, in contrast to the "lumped parameter" approach of the USLE/MUSLE. An example is Khanbilvardi's (1983) Erosion-Deposition Model (EDM).

The estimation of nutrient (N,P) loadings to receiving waters involves an additional degree of complexity. Nutrients can be received in either sediment-attached or dissolved states, and while the majority of total N and P delivered to waterways is sediment-associated, the ratio of attached to soluble nutrients is a function of land use, fertilizer composition, soil structure and chemistry, and hydrological and storm-related factors. In addition, both N and P can be delivered in a variety of chemical states. The availability of nutrients for aquatic uptake is determined to a great extent by the form in which it is delivered. For example, inorganic, dissolved P is believed to be almost 100% bioavailable, whereas particulate P is estimated to be between 20%-40% bioavailable (Nelson and Logan, 1983).

In addition, nutrients transported with sediment tend to exist in higher concentrations than in the field soils where they originate. Basic nutrient loading functions rely on estimates of sediment delivery, as described above, and employ nutrient enrichment ratios. Sediment loadings are modified to indicate nutrient loadings as follows:

$$L_{(N,P)} = \alpha \; S \; C_{s(N,P)} \; R \; E_{(N,P)}$$

where L is loading of nutrient (N,P) in mass per unit time, S is sediment loading, C_s is soil surface layer concentration of N or P, R is ratio of mean particle density of surface soil to mean particle density of sediment, and E is the enrichment ratio: mass of nutrient in sediment to mass in surface layer. The enrichment ratio is inversely related to volume of soil loss. The loading function for Phosphorous must further be adjusted to reflect the degree to which P is available for biological uptake.

The particulate form of the nutrient, specifically P, is associated with the fine soil fractions, which are more highly represented in sediment when overall sediment transport is low. As storm intensity and overland flow volume increase, the composition of sediment more closely resembles that of field soil. The heaviest soil fractions are the first to be re-deposited, and sediment reaching water bodies typically contains a concentration of fine particles like finer silts and clay. Nelson and Logan (1983) have estimated this relationship and found enrichment ratios for P ranging from 1 to 10, with values between 2 and 6 common. Other researchers have obtained enrichment ratios for total N of between 1.08 and 5.0, organic matter between 1.15 and 4.7, and biologically available P between .99 and 3.74 (Dean, 1983).

A number of models have been developed on both the field and watershed scale to allow the prediction of agricultural NPS loadings of sediment, nutrients, and in some cases pesticides to surface and ground waters. They can be classified as either lumped parameter or distributed parameter approaches. The former treats the entire watershed as a single hydrologic unit, while the latter partitions the watershed into "cells" or sub-regions of discrete and relatively uniform characteristics. Both approaches typically include an erosion-deposition component, often a modified form of the USLE, a hydrologic component, and in some cases chemical or nutrient submodels. Lumped-Parameter models include the Cornell Nutrient Simulation model (CNS) developed by Haith (1979), the Chemicals Runoff and Erosion from Agricultural Management Systems (CREAMS) model developed by Knisel (1980), and the Agricultural Runoff Model (ARM) developed by Donigian, et al., (1977).

Comprehensive watershed models, including the Watershed Evaluation and Research System (WATERS, Carvey and Croley, 1984), the Areal Nonpoint-Source Watershed Environment Response Simulation model (ANSWERS, Beasley and Huggins, 1982) and the Agricultural Nonpoint-Source (AGNPS) model, have been employed to evaluate the impact of management practices and conservation policies on NPS pollution loadings at the watershed level. These are distributed parameter hydrologic models employing a common basic approach: watersheds are divided into "cells" or subunits possessing relatively uniform characteristics (e.g., slope, soil type, and rainfall erosivity.) Erosion-deposition models (often based on the USLE) are used to calculate soil and nutrient dislocation within each cell. Cells are connected via a series of "stream tubes" which ultimately link all cells to tributaries and finally to the watershed outlet via hydrologic models.

The entire framework serves as an accounting matrix for sediment and nutrients. Each cell receives sediment and nutrients exported from the cell immediately upstream. New sediment is dislocated within the cell, and sediment in excess of overland flow capacity is re-deposited. Models are typically calibrated on watersheds for which empirical measurements of sediment and nutrient loadings have been made over various weather events.

The use of models such as the above, as well as physical monitoring of NPS pollutant loadings under different land management practices, has permitted estimation of the quantitative impacts of BMP's on water quality. The simplest modeling approach involves the C and P parameters of the USLE and the assumption that the SDR is invariant to management practices. The C (cover and management) and P (support practices) factors are under the farmer's control, and will take on lower values when land is shifted from erosive practices to BMP's. The change in the combined C and P term predicts the reduction in gross erosion associated with a shift in management practices. For example, by switching from conventional tillage with a moldboard plow to a no-till system in a corn-small grain rotation, gross soil loss is estimated to decrease by approximately 50%. Contouring is estimated to reduce erosion by 50% when replacing up-and-down cultivation on slopes of between 3% and 8%, and when combined with strip cropping can reduce erosion to 10% of pre-practice levels (Wischmeier and Smith, 1978).[8]

Clark et al. (1983) have summarized the conclusions of several studies quantifying the impact of land management practices on delivery of sediment, nutrients, and pesticides, representing a variety of locations and methodologies (Table 19). Estimated reductions for a given land

Table 19 - Effects of Land Management Practices on Delivery of Sediment, Nutrients, and Pesticides.

Land Management Practice	Percentage Reduction in the Yield of: Runoff	Gross Erosion	Sediment	Nutrients	Pesticides
Tillage Practices:					
Contouring	up to 20%		20% - 70%	25% - 65%	20% - 25%
Conservation Tillage	10% - 60%		15% - 90%	15% - 70%	uncertain (a)
Cropping Patterns:					
Rotation (b)		50% - 70%		30% - 80%	
Strip Cropping		up to 85%	75% (c)		
Structural Measures:					
Terraces	up to 90%	75% - 95%	30% - 75%	20% - 95%	up to 50%
Sediment Basins			up to 90%	approx. 50%	
Grassed Waterways	2% - 50%		25%	10% - 70%	up to 75%
Other Practices:					
Filter Strips	25%		25%	25% - 50%	
Mulching	25%		50%		
Pesticide, Fertilizer Management				up to 50%	up to 75%
IPM					20% - 40%
Irrigation Control			up to 90%		

(a) Studies have estimated reductions of up to 99%, others estimate increases in pesticide delivery. Groundwater infiltration of pesticides is often found to increase under conservation tillage.
(b) Adding small grain and hay to a row-crop rotation.
(c) In association with contour plowing.

management practices are highly variable, reflecting the influence of soil, topography, weather, and baseline assumptions.

Knisel et al. (1983) employed the CREAMS field-scale model to estimate the impact of several management practices on sediment and nutrient yields on Georgia soils (table 20). The Enrichment Ratio (ER) refers to the degree to which nutrients are concentrated in run-off sediment as compared to soil in the field. Enrichment ratios increase as mean particle size decreases (because of higher clay content in small particles), and the ratio of small to large particles increases as overall runoff decreases.

Overall, evidence from both empirical studies and simulations supports the effectiveness of agricultural BMP's in reducing upstream erosion and delivery of sediment and associated pollutants to surface waters. The extent to which these reductions translate into meaningful improvements in downstream water quality cannot be assessed, however, without further examination of the characteristics and behavior of receiving water systems.

Lake Changes from Reduced Nutrient Loading

One issue of concern is the relationship between changes or reductions in external nutrient loadings to freshwater lakes and the consequent changes in lake trophic states and water quality. This relationship can be quantified using nutrient budgets, which itemize nutrient loads, sources, and sinks within a lake. Once empirical nutrient budgets have been established for a given lake, empirical or mechanistic models can be used to predict the effect of an external nutrient load change. A comprehensive discussion of empirical models for lake trophic

Table 20 - Impact of Tillage Practice on Sediment and Nutrients.

Tillage & Management:	Sediment T/ha	(%)	ER	N kg/ha	(%)	P kg/ha	(%)
Conventional Tillage (CT)	21.21	(100)	2.1	41.4	(100)	15.64	(100)
CT + Grassed Waterways (GW)	10.75	(51)	2.7	25.8	(62)	9.64	(62)
CT using Chisel Plow, GW	4.01	(19)	2.3	11.2	(27)	4.18	(27)
CT + terraces + contour	3.85	(18)	2.9	10.8	(26)	3.98	(25)
CT + outlet impoundment	2.15	(10)	4.3	9.1	(22)	3.34	(21)

Source: Knisel et al. (1983).

status evaluation is found in Reckhow and Chapra (1983) and Chapra and Reckhow (1983).

Empirical studies reviewed by Maki, et al. (1984) and others suggest that the response of ambient lakewater quality and lake trophic status to changes in nutrient loadings from external sources are less than proportional. In reviewing several field studies of natural lakes, Maki found that reductions of external P loadings of up to 50% may not substantially improve lakewater quality. Rast and Lee (1978) also concluded that the elasticity of response of secci disk transparency and/or chlorophyll concentrations was less than 1%. Research by Bachman (1980) suggests that "...many eutrophic waters cannot be expected to significantly respond unless nutrient loading declines even more than the 50% - 90% reduction expected from some BMP's" (Menzel, 1983, p. 16). This is largely due to nutrient loads accumulated internally and nutrient recycling within lakes. Many lakes which have reached a eutrophic state contain more biologically available P internally than is delivered annually from external sources. Even if external loads were eliminated entirely (which is seldom technically feasible), nutrient recycling would keep these lakes eutrophic for years or possibly decades.

A second reason why lakewater quality responses to reduced external loading are less than proportional is the hysteresis phenomenon. Conceptually, the recovery path that an aquatic ecosystem (or a species within that ecosystem) follows once a disruptive pollutant is removed (or delivery reduced) is not a re-tracing of the disruption path. In addition, the long-term equilibrium achieved by the ecosystem following the removal or abatement of the pollutant may not resemble the pre-pollution equilibrium. Studies on lake restoration have in some cases shown that ecosystem damage and recovery are non-linear with respect to pollutant loads, and significant time lags are often observed. The non-linearity and lags are believed by some researchers to reflect the varying turnover times of the physical, abiotic, and simple biotic components of the aquatic ecosystem (Horne, et al., 1985). Perturbations of the primary producer species (e.g., phytoplankton) are amplified through the food chain, and species higher on the food chain such as game fish are affected more dramatically.

Relatively few high-quality empirical lake recovery studies have been performed, and many of the tentative observations presented here follow from model simulations. On the basis of existing evidence, however, it is reasonable to conclude that programs which improve surface water quality and sport fisheries through agricultural BMP's are most likely to succeed where (1) pollutant loadings from agriculture are the primary source of water quality problems in a given lake or watershed, <u>and</u> (2) receiving waters have not already reached an advanced eutrophic state and/or suffered game fish species displacement or elimination. When ecosystems are damaged or contain sufficient internal nutrient loadings to maintain a eutrophic state independent of changes in external loadings, lake restoration techniques must be employed.

The above also suggests that cost-effective targeting of soil conservation resources should involve identifying watersheds where implementation of BMP will act to <u>preserve</u> water quality. Where aquatic ecosystems are relatively healthy, land management practices alone may be sufficient to maintain this state. Where disruption has occurred, and restoration will be required to achieve acceptable water quality and sport fisheries, the off-site economic benefits from agricultural BMP's will be correspondingly lower. Of course, future payoffs to lake restoration activities would be enhanced by concommitant reductions in pollutant loadings from agriculture.

Critical Thresholds and Non-linearity of Benefits

The results of both economic valuation studies and research in the physical sciences suggest that economic benefits from water quality improvement are not related in a linear fashion to reductions in upland erosion. Water quality thresholds exist so far as the survival, reproduction, and predominance of desirable species of game fish are concerned. Perceptive and aesthetic water quality thresholds exist as well, although they appear to exhibit regional variation, and vary with regard to the water quality experiences of recreational users.

Consequently, any application of agricultural BMP's intended to generate real benefits to recreational water users should be effective in moving water quality across both objective and subjective thresholds. If other pollution sources contribute to quality problems in the watershed, combined abatement efforts might be necessary to achieve this goal. Only when water quality moves across a threshold determining species survival or dominance will it lead to a clear improvement in fishing success. Likewise, only when objective water quality improves to the extent that it can be perceived by recreationists will benefits from that linkage be realized. Considerable effort has been made at both the Federal and state levels to develop water quality criteria and standards that both implicitly and often explicitly recognize thresholds, primarily on the basis of objective criteria. National "fishable" and "swimmable" water quality goals, for example, can be expressed in terms of dissolved oxygen

(Figure 8). Similar parameters have been established for other uses, many of which are summarized in the U.S. EPA (1976b) "Red Book."

Figure 8 - Resources For the Future Water Quality Ladder

Dissolved Oxygen (PPM):

...2.0...2.5...3.0...3.5...4.0...4.5...5.0...5.5...6.0...6.5...7.0......

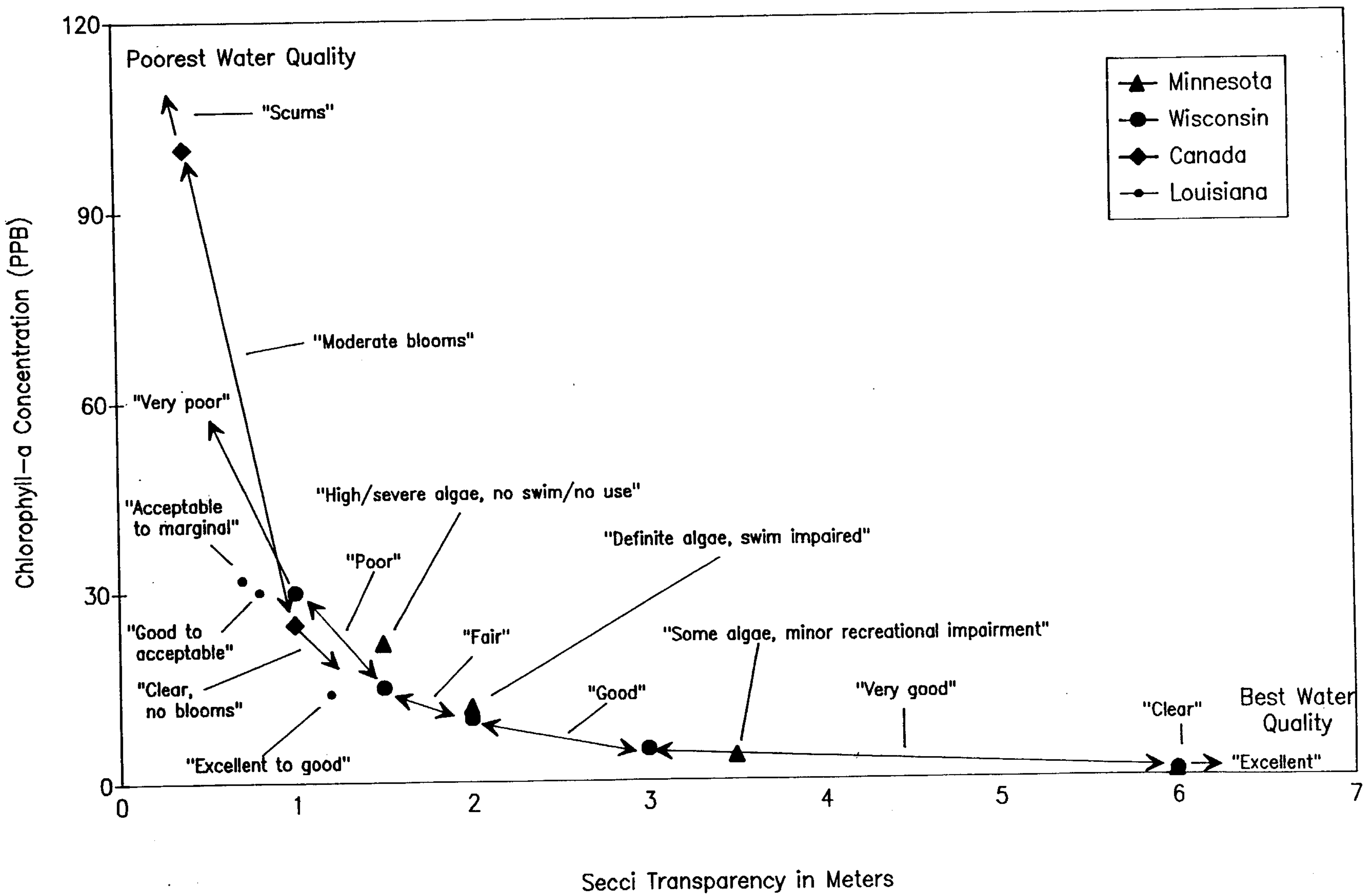
Unacceptable for Boating	Boatable	Fishable +Boatable	Swimmable +Fishable +Boatable

Source: W. T. Vaughan (1981) Appendix II

Greater difficulties arise when investigators or agencies attempt to identify analogous subjective criteria. These criteria are expected to be influenced by location, recreator experiences and expectations, and other factors militating against standardization. The problem is illustrated in Figure 9. Waters with secci transparencies of around 1 meter, and chlorophyll concentrations of 20-30 ppb are judged as "good to acceptable" and "clear" in two studies (Louisiana, Canada) but "poor to very poor" and "high algae/no swimming" in two others (Wisconsin, Minnesota) where recreator expectations of water quality appear to be higher.

An alternative approach to establishing subjective threshold values is to focus on the degree of reduction. Lee and Jones (1986) analyzed data on over 400 lakes in the U.S. and Europe and have proposed a general relationship between changes in external P loadings and detectable changes in water quality. "... a reduction of approximately 20% in the phosphorous load to a water body must be achieved to produce a discernable effect on water quality. This 20% value is independent of the trophic state" (Lee and Jones, 1986, p. 330-331). By observing this 20% rule in

Figure 9 – Regional Variation in Subjective Judgements of Water Quality



addition to targeting water quality changes around objective thresholds (where they can be identified,) programs to reduce agricultural NPS pollution will have a greater chance of generating significant off-site benefits.

CONCLUSIONS AND RECOMMENDATIONS

Two broad conclusions appear justified on the basis of the above literature review. First, the existence of tangible and substantial willingness-to-pay for water quality improvement associated with soil conservation has been documented empirically. By explicitly reflecting these values (including regional and other systematic variations) through resources targeting, soil conservation programs will achieve greater economic efficiency. Second, variation in the quantitative benefit estimates, in the research methodologies and in the assumptions of existing empirical studies does not allow an identification of regionally determined variations in water quality benefit estimates at a useful level of confidence. In particular, the studies concerned with agricultural nonpoint-source pollution produce an extremely divergent range of estimates.

A first step toward narrowing this range would involve the formal "meta-analysis" of existing empirical studies of water quality demand. The approach of Smith and Kaoru could be expanded to include classes of explanatory variables for general valuation methodology (CVM, TC, Discrete Choice, and so on) and type and extent of water quality change associated with each study. The success of such an approach would rest, minimally, on (a) having access to the raw data used in each available study, and (b) the development of a framework within which disparate measures of water quality and water quality change could be compared meaningfully.

This leads directly to a second topic of inquiry where existing knowledge is inadequate: the measurement of water quality. The successful application of the RFF Water Quality Ladder in CVM studies underscores the importance of identifying the proposed water quality change or goal with explicit reference to use, and additionally to linking specific uses to specific objective measures of quality. A problem arises in that "suitability for use" is both objectively and subjectively determined. Several states for which water-based recreation is economically important (e.g., Minnesota and Wisconsin) have developed or are developing suitability-for-use criteria that reflect both recreator perceptions and tastes and objective water quality parameters at the state level.

A third research approach with potentially high pay-off involves the linkage of economic and physical models. Examples were discussed above and include AGNPS linked to economic models (Kozloff, Minnesota/EPA). Data requirements are extensive, but high quality physical data is increasingly available as a consequence of the NRI and RCA, as well as state-level data collection efforts.

Existing information is sufficient to allow the evaluation of soil conservation projects targeted to improve surface water quality, even without precise estimates of off-site recreation estimates. This could be done by establishing criterion to screen proposed soil conservation projects or to assist in the regional targeting of agency resources. One such criteria would involve any project expected to yield significant water quality benefits to pass the following three tests:

(1) The project must be capable of moving water quality across at least

one objective threshold, such as from "non-boatable" to "boatable" water, or from non-game-fishable to game-fishable water.

(2) The project must reduce downstream deliveries of sediment and nutrients by at least 20%.

(3) Projects that serve to protect water that is not already degraded are to be selected over those which propose to improve water of poor quality. Available evidence suggests that reduced downstream deliveries, even as low as 10% of pre-project levels, may not result in any significant improvement in downstream water quality if downstream waters are already eutrophic. Any projects that propose to improve the quality of such degraded waters must explicitly address the need to include the costs of lake restoration on the cost side of the equation.

NOTES

1. For an expanded list of off-site damages, see E.H. Clark II et al. (1985), Eroding Soils: The Off-Farm Impact. Washington, D.C.: The Conservation Foundation and American Agricultural Economics Association, Soil Conservation Policy Task Force (1986), Soil Erosion and Soil Conservation Policy in the United States, Occasional Paper #2.

2. See e.g., P. Crosson and A. Stout (1983), Productivity Effects of Cropland Erosion in the United States. Washington, D.C.: Resources For The Future, and R. Strohben (Ed.) (1986), An Economic Analysis of USDA Erosion Control Policies: A New Perspective. Washington, D.C.: USDA/ERS/NRED Agricultural Economics Report # 560.

3. Based on findings presented in L.P. Gianessi and H.M. Peskin (1981),"Analysis of National Water Pollution Control Policies 2: Agricultural Sediment Control," Water Resources Research 17(4) p.804.

4. The Reagan administration's "Principles and Standards" specifically identifies economic efficiency as the criterion guiding Federal policy, and benefit-cost analysis as the framework within which efficiency is defined. Kozloff (1989) has established the basis for targeting as a policy instrument in both the theory of environmental control, and the economics of information. K. Kozloff, "Micro-Targeting Nonpoint Pollution Control: Theory and Empirical Issues." University of Minnesota, Department of Agricultural and Applied Economics, Draft First Quarter Report to US EPA.

5. Benefit estimates emerging from hedonic property value studies cannot be standardized in a manner that permits direct comparison with figures generated by recreational demand models (Tables 8, 9 and 10).

6. Appendix A identifies the source of each water quality index and the manner in which it has been interpreted for purposes of comparison. The appendix also describes how baseline water quality and water quality changes were defined for each of the studies which did not contain explicit or objective descriptions of water quality. The procedures used to accomplish this are also necessarily ad hoc, but are unavoidable if comparisons between studies are desired.

7. Contingent Behavior studies are here classified as direct methods: although CS estimates are established using a TC procedure, the change in CS associated with water quality change is established through responses to hypothetical questions rather than through observed transactions.

8. The calculations are based on tables 5, 14 and 15 from Wischmier and Smith (1978). Estimates for reductions associated with no-till are simple averages over 7 crop stages for common values of stubble residue and weather events. In practice, the C factor of the USLE should be calculated over an entire rotation to obtain a more accurate estimate of soil loss reduction.

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APPENDIX: KEY TO WATER QUALITY MEASURES, AND DETERMINATION OF WATER QUALITY CHANGE FOR INDIVIDUAL STUDIES:

I) Water Quality Measures:

The RFF Water Quality Ladder (WQL): Vaughan, W.J. (1981) "The Water Quality Ladder", in C. S. Russell and V. K. Smith (1988) Demands For Data and Analysis Induced by Environmental Policy. North Carolina State University Faculty Working Paper # 122, p. 65. The WQL is keyed to Dissolved Oxygen (DO).

"Nielson" is based on Vaughan, W.J., C.S. Russell, L.P. Gianessi, and L.A. Nielson (1982) "Measuring and Predicting Water Quality in Recreation-Related Terms", J. Environmental Economics and Management 15 p. 369 Table 1 and p. 371 Table 2. The fisheries classification system, based on research by Nielson, also depends on ambient water temperature, pH, and TSS.

The Lake Condition Index (LCI) was developed by Uttormark, P., and P. Wall (1975) Lake Classification - A Trophic Characterization of Wisconsin Lakes. Corvallis, Ore.: U.S. Environmental Protection Agency Environmental Research Laboratory. The LCI is based on Secci transparency, chlorophyll concentration, DO, and extent of winter fish-kill. A departure from the "no impairment" value for each factor is assigned "penalty points" which are summed to form the LCI, in which 0 indicates best quality and 23 worst quality. The scale appearing in Figure 5 is based on DO and Secci transparency components only.

Secci Transparency has been reconciled with D.O. based on judgments used in constructing the LCI. A secci transparency of greater than 7 m. is assigned 0 penalty points in the LCI which is here interpreted as "no impairment" and aligned with WQL "swimmable" (D.O. = 7 ppm.) The maximum number of penalty points are associated with secci transparency <.5 m, indicating significant impairment, so transparency <.5 m is here aligned with "non-boatable." Points in between .5 m and 7 m are located using the interpolation equation: $DO = 2.987\ln(secci+1) + .79$.

The Water Quality Index (WQI) was developed by Huang, C. (1986) in his Ph.D. Dissertation The Recreation Benefits of Water Quality Improvement in Selected Lakes in Minnesota, University of Minnesota Department of Agricultural and Applied Economics. It is based in Horton, R.K., (1965) "An Index Number System for Rating Water Quality", J. Water Pollution Control 37(3) pp. 300-306. Huang's WQI incorporates DO, alkalinity, pH, and Secci disc transparency. The scale appearing in Figure 5 is based on DO and Secci components only, weighted 2:1.

The "Descriptive" category is based on the LCI and is aligned with D.O. on the same basis used to align secci transparency: "No algae (bloom)" indicates no impairment (swimmable) and "regular, heavy algae bloom, mats" corresponds to maximum impairment (non-boatable.)