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The Economics of Total Maximum Daily Loads

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ABSTRACT

This article begins by exploring the institutional and historical forces shaping the total maximum daily load (TMDL) approach from an economic perspective. Next, it discusses the suitability of applying various types of economic analyses and policy instruments to individual TMDLs or the TMDL program. Cost-effectiveness analysis is shown to be particularly amenable to and appropriate for application to individual TMDLs while pollution trading may offer a mechanism for achieving water quality goals at lower cost. The final section of the article presents a case study that documents a formal application of economic analyses to an actual TMDL.

INTRODUCTION

The role and relevancy of economic analysis as a tool in total maximum daily load (TDML) allocation and implementation is receiving growing attention. In rapid fashion, the TMDL program has become a premiere water quality program in the United States while the field of environmental economics has blossomed. These combined forces have promoted greater public and governmental consideration of integrating cost-effective and market-based solutions into TMDL deliberations. Nonetheless, economic analysis is rarely incorporated into TMDLs at more than a cursory level. Thus, there is considerable potential for improvement in the economic performance of TMDLs.

TMDLs are numerical statements of maximum pollutant loads that are deemed consistent with a water body's designated uses. Section

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303(d) of the Clean Water Act[1] stipulates that states must establish TMDLs for those waters not meeting applicable water quality standards. States are subsequently required to develop implementation plans detailing how TMDLs can be achieved. Allocation of pollutant loads among dischargers involves important economic efficiency implications. In most cases, there are alternative strategies and load allocations that will achieve a TMDL. Failure to choose an economically efficient strategy means that a higher cost than necessary will be paid to achieve a TMDL, thus placing undue economic burden on society.

This article begins by exploring the institutional and historical forces shaping the TMDL approach from an economic perspective. The suitability of applying various types of economic analyses and policy instruments to individual TMDLs or the TMDL program, generally, is next discussed. The final section of the article presents a case study documenting a formal application of an economic analysis to an actual TMDL.

AN ECONOMIC PERSPECTIVE OF THE CLEAN WATER ACT

Since 1972, the landmark Clean Water Act (CWA) and subsequent amendments have dominated the institutional landscape of water quality regulation in the United States. Although not completely absent, references to costs or benefits in the CWA are few. The major regulatory mechanism of the CWA, the National Pollutant Discharge Elimination System (NPDES), focused on the technical feasibility of achieving effluent standards as the primary factor in establishing pollution controls in contrast to former clean water legislation, with its emphasis on state-administered ambient standards.

During debate of the 1972 amendments, classic economic arguments were put forth in favor of ambient-based water quality legislation.[2] Countering this line of argument was a consensus, which

1. Federal Water Pollution Control Act, 33 U.S.C. §§ 1251-1387 (2000). In 1977, Congress formally recognized the common nomenclature, "Clean Water Act." The 1972 Clean Water Act and subsequent amendments are collectively referred to as the CWA or "the Act" in this article.

2. For example, in House testimony, the Chairman of the Council of Economic Advisors (CEA) stated,

> CEA is in agreement with the use of water quality targets appropriate to the conditions and expected uses of water in particular areas of the country. That is basic to the concept of relating the costs of programs to the benefits received from them. To abandon that concept for a nationally legislated standard which focuses on the level of pollutants removed and is unrelated to water quality uses and standards is economically unwise because it means a necessary misallocation of our inevitably scarce economic resources.

ultimately prevailed, that the former state-based ambient approach had been given ample time to succeed but had overwhelmingly failed.[3] Apart from differences in scientific approaches, perhaps the strongest argument in favor of NPDES, versus the former state-based legislation, was that only strong federal control could force states to cleanse their waters. Thus, despite some economic and scientific misgivings of the prescriptive NPDES approach, it was at least viewed as an approach that could be effectively implemented and would actually produce cleaner water, the benefits of which were, presumably, greater than the administrative and social costs of the program.[4] Yet despite the committed implementation of NPDES, standards were often not met, due in large part to unregulated nonpoint source pollution.[5] Although section 303(d) of the CWA (the legislation establishing the TMDL program) was largely ignored for the first two decades after passage of the Act, a series of lawsuits starting in the 1980s forced both the states and the U.S. Environmental Protection Agency (EPA) to prepare for an impending tidal wave of TMDL activity. In 2001, state lists indicated about 21,000 impaired water bodies requiring in excess of 40,000 TMDLs,[6] and these numbers have continued to grow. These figures suggest that the TMDL process will continue as a keystone water quality control program for decades to come.

Was the abandonment of ambient standards irrational? Is the return to ambient standards in the form of TMDLs a prescription for inaction? I will argue that these developments in water quality policy can be viewed as progress toward a more economically rational and effective system of water quality control. Two reasons often provided for the failure of pre-1972 ambient-based water quality legislation are (1) the

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Oliver A. Houck, *TMDLs: The Resurrection of Water Quality Standards-Based Regulation Under the Clean Water Act*, 27 ENVTL. L. REP. 10,329, 10,334 (1997) (citing *Water Pollution Control Legislation—1971: Hearings on H.R. 11,896, H.R. 11,895: Before the Comm. on Public Works, 92d Cong. 483 (1971)).*

3. *See, e.g., id.* at 10,335.

4. The interjection of economic (or cost) considerations into CWA deliberations, in fact, was viewed as anathema by environmentalists and a large portion of the American public, thus adding to the appeal of the more direct NPDES approach. Economist Wallace Oates recounts, "Environmentalists...flatly rejected an economic approach (as I learned personally and painfully on several occasions) and called for direct controls on polluting activities." Wallace E. Oates, *Forty Years in an Emerging Field*, 137 RESOURCES 8, 9 (1999).

5. It is important to note, however, that the NPDES program is overwhelmingly regarded as a success in controlling point source pollution and that it was never intended to control nonpoint sources.

6. NAT'L RESEARCH COUNCIL COMM. TO ASSESS THE SCIENTIFIC BASIS OF THE TOTAL MAXIMUM DAILY LOAD APPROACH TO WATER POLLUTION REDUCTION, ASSESSING THE TMDL APPROACH TO WATER QUALITY MANAGEMENT, 2 (2001).

inadequate state of science and technology[7] and (2) the unwillingness or lethargy of states in implementing and enforcing state standards. Neither of these conditions, however, implicates the ambient-based approach per se or in principal, when the conditions are ripe.[8] State inaction was addressed by the partial federalization of water pollution control,[9] and, although still challenged,[10] the nation is clearly better prepared today than 30 years ago to take on the enormous scientific endeavor required to support TMDLs.[11]

Economists acknowledge that techniques for calculating costs and benefits in a real world setting were inadequately developed at the time of the CWA deliberation.[12] The relative immaturity of the environmental sciences and environmental economics, thus, provide rationale for the subordination of the ambient-based water quality approach in favor of effluent standards. While economic theory might rate the ambient approach as a "first best" solution, the theoretically "second best,"[13] effluent approach, was viewed as a solution that at least

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7. It is now widely acknowledged that the data, science, and analytical capabilities needed to support ambient-based water quality control for thousands of interconnected waterways were lacking in the 1960s and early 1970s, *e.g.*, a committee appointed by the National Science Foundation concluded, "The 303d focus on ambient water quality standards has returned the nation to a water quality program that was not considered implementable 35 years ago when there was a paucity of data and analytical tools for determining causes of impairment and assigning responsibility to various sources." *Id.* at 16.

8. Tietenberg concludes that "the wrong inference was drawn from the early lack of legislative success [in achieving water quality]." THOMAS H. TIETENBERG, ENVIRONMENTAL AND NATURAL RESOURCE ECONOMICS 489 (3d ed. 1992).

9. Although the EPA was given ultimate responsibility for enforcing TMDLs or conducting them if states failed, states were required to list waters and conduct and implement TMDLs. Some analysts are skeptical that this partial federalization will work. *See, e.g.*, Oliver A. Houck, *TMDLs III: A New Framework for the Clean Water Act's Ambient Standards Program*, 28 ENVTL. L. REP. 10,415, 10,436 (1998).

10. A recent GAO report seriously questions whether the data and science needed to conduct TMDLs are adequate even today. A mail survey sent to water quality administrators of 50 states indicated that a majority of states lacked the data they needed to manage ambient water quality. GEN. ACCOUNTING OFFICE, PUB. NO. GAO/RCED-00-54, WATER QUALITY: KEY EPA AND STATE DECISIONS LIMITED BY INCONSISTENT AND INCOMPLETE DATA 44 (2000); *see generally id.*

11. While recognizing lawmakers' concern over the "paucity of data and information available to the states...[to] meet water quality standards," a committee appointed by the National Research Council concluded that "the data and science have progressed sufficiently over the past 35 years to support the nation's return to ambient-based water quality management." NAT'L RES. COUNCIL, *supra* note 6, at 2, 3.

12. *See* Oates, *supra* note 4, at 9.

13. A first best solution maximizes social welfare but ignores transactions costs and implementation issues. A second best solution maximizes social welfare under a constraining set of social or administrative limitations.

could be effectively implemented, given the existing development of the sciences and technology.[14]

A final argument can be added in support of the economic rationality of focusing on effluent standards first, then turning to ambient standards after effluent standards have been fully implemented. Apart from implementation costs, the time, effort, and expertise required to conduct individual TMDLs "worth their salt" are undoubtedly more costly than promulgating and enforcing across-the-board national effluent standards.[15] Taking all factors into consideration, particularly transactions costs, a simple, across-the-board, low-cost system can be more cost-effective than a complex site-specific approach.[16] In cases where additional progress is required, however, a site-specific approach allowing greater flexibility of design and implementation may be required.[17] By way of analogy, ordinary detergent, generally applied, gets the bulk of the wash clean, while spot remover and additional scrubbing are reserved for the tough stains. Thus, while a variety of views exists regarding the rationality and effectiveness of CWA legislation,[18] there is considerable support for the premise that the initial abandonment and subsequent return to ambient standards in the form of TMDLs was, in the main, very consistent with economic rationality.

AN ECONOMIC CHARACTERIZATION OF WATER QUALITY

Water pollution is a classic externality produced by various types of economic activity. An industrial facility, for example, may

14. Ackerman and Stewart, while strongly supporting market incentives, concede that the prescriptive Best Available Technology (BAT) approach "made some sense as a crude first-generation strategy." Bruce A. Ackerman & Richard B. Stewart, *Reforming Environmental Law: The Democratic Case for Market Incentives*, 13 COLUM. J. ENVTL. L. 171, 199 (1988).

15. The costs, difficulties, and complexity of conducting as well as implementing TMDLs is often cited as a top concern in assessing the wisdom of heading down the TMDL path. *See, e.g.*, Tara Hun, *Costs, Nonpoint Sources Are Top TMDL Concerns*, 10 WATER ENV'T & TECH. 36, 36-37 (1998). An EPA study of development costs for 14 TMDLs reported total costs ranging from $4039 to $1,024,000. ENVTL. PROT. AGENCY, PUB. NO. EPA-R-96-001, TMDL DEVELOPMENT COST ESTIMATES: CASE STUDIES OF 14 TMDLS 12-14 (1996).

16. Providing some economic rationale for simple rules, economist Davidson writes, "I think the simple rule of requiring everyone to drive on the right side of the road results in a fairly optimal system, and offhand I cannot suggest a cost incentive system that would be as effective and simple. Hence we should not neglect the possibility of simple rules in our search for offsetting some types of externalities." Paul Davidson, *The Valuation of Public Goods*, *in* ECONOMICS OF THE ENVIRONMENT: SELECTED READINGS 345, 354 n.12 (Robert Dorfman & Nancy S. Dorfman eds., 1972).

17. Boyd points out, "The low-hanging fruit of low-cost high-volume point source controls has been harvested. Today, significant water quality improvement requires the expansion of controls to nonpoint sources." Jim Boyd, *Unleashing the Clean Water Act: The Promise and Challenge of the TMDL Approach to Water Quality*, 139 RESOURCES 7, 8 (2000).

18. *See, e.g.*, Houck, *supra* note 2; NAT'L RESEARCH COUNCIL, *supra* note 6.

discharge toxic effluent into a river causing fish kills and environmental impairment downstream. Because the fishermen and recreationists that suffer harm are not parties to the plant's decision to discharge, that harm is external to the decision process. Another key feature of water pollution is that, in almost all cases, it harms multiple users and nonusers simultaneously and, hence, is a *public* bad. From the reverse perspective, clean water is a public good producing benefits to large groups of users but requiring costs for its attainment. The externality and public goods nature of water quality, as for other environmental media (air, land), often lead to market failure: a situation where exclusive reliance on market forces does not result in socially desirable outcomes. The major causes for market failure in water quality are undefined water quality rights and control obligations and the practical obstacles that discourage numerous individuals, receiving relatively modest water quality benefits, from coordinating their efforts and directly contracting with dischargers. Societies have sought to correct market failures through various forms of social intervention.[19]

COMPONENTS OF ECONOMIC ANALYSIS

Two fundamental building blocks of applied microeconomic analyses are costs and benefits. Water pollution control involves economic sacrifices or costs for some groups (mainly the owners of production or industry) while benefiting others (generally, the public). Alternatively, water pollution imposes costs on the public while providing benefits to industry. The following sections discuss costs and benefits from a water quality versus a water pollution perspective, *e.g.*, costs refer to the control of attaining water quality while the benefits refer to the benefits derived from water quality.

Costs

Due to constraints imposed by nature and technology, producing more of one good, including water quality, is typically accomplished only at the expense of producing less of other goods. Occasionally, analyses suggest that water quality control measures may also result in net benefits to dischargers. Such "win-win" solutions are the exception but might occur if (1) current technology and resources are not being efficiently utilized or (2) recent technology has expanded opportunities for improving water quality while simultaneously

19. While not exhaustive, alternative externality correction devices include market emergence, merger, economic incentives, regulation, prohibition, pseudo markets, and moral suasion.

producing economic benefits. Numerous water pollution control measures have been identified for municipalities, industry, and agriculture. The costs of implementing these measures are often estimated and typically expressed in monetary units, *e.g.*, dollars, which represent the foregone opportunity of producing and consuming other goods.

Benefits

The benefits of water quality can be classified as environmental and economic. Environmental benefits are measured by indices such as biological diversity, increased fish populations, etc. Economic benefits are the monetary values society places on water quality benefits, often measured by society's willingness to pay for the associated environmental benefits. Water quality benefits include improvements in (1) recreation (swimming, fishing, boating, water fowl hunting), (2) nonuser benefits (amenity, aesthetic, and ecological benefits not directly associated with activities on or near a water body but for which households may be willing to pay), (3) diversionary uses (reducing risk to human health and decreased costs for municipal water supplies), and (4) commercial fisheries.[20] Many of the benefits listed above are nonmarket, *i.e.*, they are not purchased or sold and therefore have no observable price. The lack of organized markets for water quality complicates the estimation of water quality benefits, though a number of indirect methods have been developed.

ECONOMIC INSTRUMENTS FOR ANALYZING WATER QUALITY POLICY

Cost-benefit analysis and cost-effectiveness analysis are two basic but well-worn tools in the economic analysis toolkit and both have been applied extensively to water quality issues.

Cost-Benefit Analysis[21]

Applied to water quality, cost-benefit analyses generally estimate and compare both the economic costs and benefits of a water quality initiative. Such analyses are, by their nature, prima facie

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20. This etymology is based on A. MYRICK FREEMAN, III, AIR AND WATER POLLUTION CONTROL: A BENEFIT-COST ASSESSMENT 9 (1982).

21. A general review of cost-benefit analysis and its applicability to environmental regulation is provided in Scott Farrow & Michael Toman, *Using Benefit-Cost Analysis to Improve Environmental Regulations*, 41 ENV'T 12 (1999).

prescriptive, *i.e.*, benefits exceeding costs could be considered a sufficient condition for proceeding with a project.[22] In theory, the optimal level of ambient water quality is such that the benefits produced by one additional unit of water quality equal the costs for its attainment. Thus, an assessment of the merits of a water quality project requires detailed information on both its benefits and costs. Significant difficulties, limitations, and complications inherent in environmental benefit estimation, described later in this article, suggest one reason why cost-benefit analysis, although theoretically appealing, has not factored more prominently in the water quality policy arena and why such analyses, when pursued, are seldom decisive.[23]

Four issues hamper cost-benefit analysis, particularly as applied to environmental quality. First, in cases where benefits cannot be adequately quantified because of uncertainty or other technical hurdles, benefits can only be described. By contrast, estimating the *costs* of structural or managerial pollution control measures is relatively straightforward and generally involves much less uncertainty. Comparing quantified costs to descriptive, though potentially large, benefits can unduly discount benefits.[24] Second, the use of cost-benefit

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22. A more exacting and universally accepted welfare economic criterion is the Pareto criterion, which states that everyone should be made at least as well off while nobody is made worse off. A policy producing net benefits (total benefits greater than total costs) is considered a *potential* Pareto improvement since, in theory, the gains of the policy action can be redistributed such that everyone is made at least as well off as before. In practice, a strict Pareto improvement is difficult to achieve although efforts are often made to compensate losers. Thus, a potential Pareto improvement, *i.e.*, the generation of positive net benefits, is often used as a theoretical decision rule.

23. It is noteworthy that authors of environmental economics textbooks, for the reasons cited, often exhort students to be cautious in their interpretation of cost-benefit analyses. *See, e.g.*, JAMES R. KAHN, THE ECONOMIC APPROACH TO ENVIRONMENTAL AND NATURAL RESOURCES 120 (2d ed., 1998) ("Although cost-benefit analysis, with its reliance on numbers and economic theory, may seem as if it might be a precise science, this is not the case."); *but see* Hartwick and Olewiler, who maintain,

> Cost benefit analysis has been maligned by those who say that it is useless because the numbers are so bad. This criticism misses the point of the exercise....The estimate of the benefits of environmental control is crucial if we are to evaluate the tremendous costs of reducing pollution and make informed judgments about the social value of improving environmental quality.

JOHN M. HARTWICK & NANCY D. OLEWILER, THE ECONOMICS OF NATURAL RESOURCE USE 432 (1986). A large burden is lifted off the shoulders of cost-benefit analysis by characterizing it as a useful tool, rather than "a singleminded decision rule." Farrow & Toman, *supra* note 21, at 37.

24. While broadly supporting an expanded use of cost-benefit analysis in governmental decision making, a group of highly respected economists also highlighted some of its limitations. One of many caveats was that "[c]are should be taken to ensure that quantitative factors do not dominate important qualitative factors in decisionmaking."

analysis is often perceived to be subject to manipulation, or at least highly subject to the analysts' biases, especially when gaps in knowledge and sources of uncertainty are substantial.[25] Some studies, for instance, have indicated that the non-use or existence value of environmental amenities far outweighs their commercial and use value, while many studies omit the estimation of these values altogether. Third, full-blown cost-benefit analyses can be very expensive and time consuming. Because of the site-specific nature of water quality benefits and costs, it would cost billions of dollars to model them, even when many benefits cannot be estimated adequately. Finally, for individual TMDL analyses, environmental criteria rather than social valuations trigger TMDLs. While considering the costs of TMDL implementation measures is permitted, even endorsed, estimating benefits has no prima facie relevancy within the TMDL framework.

Nonetheless, society must eventually decide if a project or program is worth its cost. In recent years, Congress has called for a greater use of cost-benefit analysis to justify major programs.[26] Under regulatory review provisions of Executive Order 12866,[27] President Clinton required the EPA to evaluate the costs and benefits of proposed new rules. While the Executive Order strictly applies to direct costs incurred by states to implement proposed changes in the TMDL rule, EPA has also investigated the costs and benefits to all parties resulting from implementation of the TMDL program.[28] In state water quality

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KENNETH J. ARROW ET AL., AM. ENTER. INST., BENEFIT-COST ANALYSIS IN ENVIRONMENTAL, HEALTH, AND SAFETY REGULATION: A STATEMENT OF PRINCIPLES 2 (1996).

25. Recounting his experience as the senior economist for environmental and resource policy at the Council of Economic Advisors, Jason Shogren comments: "But now I understand that some people see our orthodoxy as not just simply confining but as downright prehistoric....Because after all, cost-benefit analysis is just naked self-interest dressed up in banker's pajamas, isn't it?" Jason F. Shogren, *Do All the Resource Problems in the West Begin in the East?*, 23 J. AGRIC. & RESOURCE ECON. 309, 311 (1998).

26. ARROW ET AL., *supra* note 24, at v.

27. Exec. Order No. 12,866, 58 Fed. Reg. 51,735 (Sept. 30, 1993).

28. Proposed rules elaborate:

> In anticipation of the interest of diverse stakeholders, EPA has begun work to gather information about the costs and benefits that can be expected to result from implementation of the TMDL program. A key part of this assessment is to better understand the costs and benefits of the existing TMDL program, as well as the incremental costs and benefits that will result from the changes to the TMDL program.

Proposed Revisions to the Water Quality Planning and Management Regulation, 64 Fed. Reg. 46,012, 46,042 (Aug. 23, 1999) (to be codified at 40 C.F.R. pt. 130) [hereinafter Propposed TMDL Rule]. The final rule was due to become effective on April 30, 2003; however, on December 27, 2002, EPA announced its intent to withdraw the proposed rule and to issue a new rule in April 2003.

reports, proposed TMDL rules also require estimating the costs and benefits of measures needed to achieve CWA objectives.[29]

A committee assembled by the National Research Council to assess the scientific basis of the TMDL approach to water quality reduction suggested that estimating the economic benefits of pollution reduction in relation to its costs would also be constructive in developing appropriate use attainability analyses (UAAs) for individual water bodies. States prepare such analyses to inform the process of assigning designated uses to water bodies. In turn, water quality criteria are designed to be protective of designated uses. UAAs, therefore, are necessary precursors to assessing and listing waters and conducting TMDLs. Until recently, however, little attention has been directed toward developing appropriate UAAs. Addressing this shortfall, the committee stated, "States should develop appropriate use designations for waterbodies in advance of assessment and refine these use designations prior to TMDL development."[30]

In summary, cost-benefit analysis provides a formal mechanism to determine whether a project or a program is worth its cost. Unfortunately, estimating water quality benefits, as with other types of environmental benefits, is difficult and controversial. Moreover, estimating economic benefits has no regulatory relevancy for individual TMDLs. Nonetheless, at a broader programmatic level, EPA is investigating the costs and benefits of the entire TMDL program, and evaluation of both costs and benefits is also recommended for UAAs.

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29. "Each such report shall include...an estimate of the environmental, economic and social costs and benefits needed to achieve the objectives of the CWA and an estimate of the date of such achievement." *Id.* at 46,047.

30. COMMITTEE TO ASSESS THE SCIENTIFIC BASIS OF THE TOTAL MAXIMUM DAILY LOAD APPROACH TO WATER POLLUTION REDUCTION, NAT'L RESEARCH COUNCIL, *supra* note 6, at 4. The Committee elaborates:

> Appropriate use designation for a state's waterbodies is a policy decision that can be informed by technical analysis. However, a final selection will reflect a social consensus made in consideration of the current condition of the watershed, its predisturbance condition, the advantages derived from a certain designated use, and the costs of achieving the designated use.

Id. at 24.

Estimating Benefits[31]

At least four linkages must be established and quantified to estimate the economic benefits of water quality improvements: (1) the link between water quality measures and impairment,[32] (2) the link between impairment and environmental benefits, (3) the translation of environmental benefits into economic benefits, and (4) the link between control measure implementation and water quality measures.

For example, phosphorus reduction technology at wastewater treatment plants (a control measure) lowers discharge of phosphorus into the receiving water body (*e.g.*, a lake). This reduces ambient phosphorus concentrations (a water quality measure). Reducing the ambient phosphorus concentrations may reduce the incidence of algal blooms, which depletes dissolved oxygen (an impairment). Avoiding oxygen depletion, in turn, prevents fish kills (an environmental benefit). The final step in economic benefits estimation involves determining the level at which society values the avoidance of fish kills and other improvements associated with reductions in algal blooms.[33]

Estimating the relationships between each of these linkages presents significant scientific challenges. The effects of a control measure on a water quality indicator are often estimated by sophisticated mechanistic water quality models. These models require specialized training and many man-hours of labor for inputting data; calibrating, validating, and running the model; and interpreting results. Equally sophisticated models with biological components are sometimes employed to estimate more direct impairment measures such as elevated algal growth, as well as actual environmental benefits such as increased fish populations or biological diversity. The final (and often most controversial) step assigns a monetary value to the resulting economic benefits.

31. General reviews of benefits estimation techniques are provided by APOGEE RESEARCH, INC., U.S. ARMY CORP OF ENGINEERS, PUB. NO. IWR REPORT 96-R-24, MONETARY MEASUREMENT OF ENVIRONMENTAL GOODS AND SERVICES: FRAMEWORK AND SUMMARY OF TECHNIQUES FOR CORPS PLANNERS (1996); FREEMAN, *supra* note 20, at 8-25; and in resource and environmental economics textbooks, *e.g.*, KAHN, *supra* note 23; HARTWICK & OLEWILER, *supra* note 23; TIETENBERG, *supra* note 8. A more technical review is provided in MARC O. RIBAUDO & DANIEL HELLERSTEIN, U.S. DEP'T OF AGRIC., PUB. NO. TB-1808, ESTIMATING WATER QUALITY BENEFITS: THEORETICAL AND METHODOLOGICAL ISSUES (1992).

32. Water quality measures are not necessarily directly related to impairment. Elevated levels of ambient phosphorus in surface waters, for instance, result in impairment only under certain conditions, *i.e.*, when phosphorus is the limiting nutrient. On the other hand, some water quality measures, such as heavy metals and ammonia, more directly measure impairment.

33. Algal blooms in surface waters can also cause unpleasant odor, interference with boating and other water sports, reduced visual aesthetics, and unpleasant tasting drinking water.

Where water pollution causes direct damage to a market commodity resulting in either a decrease in production (*e.g.*, fisheries) or an increase in expense (*e.g.*, increased treatment cost for municipal drinking water), calculating the value of water quality improvements is relatively straightforward. Unfortunately, many water quality benefits are not so easily valued because they are both nonmarket and public and, hence, do not reveal a market price.

Economists have developed a number of indirect methods for estimating nonmarket values of water quality improvement. The hedonistic technique is often used to estimate a lower bound on amenity or aesthetic value. This technique infers the value of a nonmarket commodity by analyzing the value of a commodity whose value is influenced by the nonmarket commodity, *e.g.*, comparing waterfront property values across regions experiencing different levels of water pollution. Differences in human behavior, *e.g.*, increased recreation due to water quality improvements, can also be used in conjunction with travel cost models to estimate the value of recreational water quality benefits. The travel cost model assumes that the willingness to pay for a recreational trip is at least equal to the cost incurred in traveling to the recreational site. Hedonic and travel cost models are revealed preference methods because their estimation techniques depend on observable (revealed) behavior.

If values for water quality are not closely related to use, *e.g.*, the desire to provide clean water environments to current and future generations, they are not necessarily revealed through either changes in market prices or by other aspects of human behavior, such as travel or home location. Thus, observational methods alone may not reveal the totality of nonuse valuations. Researchers have circumvented this hurdle by describing hypothetical markets to respondents who are asked to indicate their willingness to pay for a nonmarket commodity. This relatively recent methodology has come to be known as the contingent valuation (CV) method, because willingness to pay values are contingent upon the particular hypothetical market described.[34] Since the situation is hypothetical and an actual transaction or observable behavioral pattern has not been affected, value estimates based on surveys are controversial. Participants may answer strategically, e.g., may indicate

34. *See generally* ROBERT CAMERON MITCHELL & RICHARD T. CARSON, USING SURVEYS TO VALUE PUBLIC GOODS: THE CONTINGENT VALUATION METHOD (1989) (often considered "the Bible" of CV analysis by its practitioners); a good summary is provided in JOSEPH BREEDLOVE, NATURAL RESOURCES: ASSESSING NONMARKET VALUES THROUGH CONTINGENT VALUATION (CRS REPORT FOR CONGRESS, PUB. NO. RL30242, 1999); the CV method has also been referred to as the survey method, the interview method, the direct interview method, the direct questioning method, the hypothetical demand curve estimation method, the difference-mapping method, and the preference elicitation method.

too high a value because they think it will improve the chance that a water quality improvement will be made, or indicate too low a value if they think their reported willingness to pay might result in an increase in taxes.[35]

The public goods nature of water quality improvements complicates the analysis and adds to data collection expense.[36] The value of water quality improvements is typically spread across a wide population, and researchers must determine the relevant population for a particular improvement. Another obstacle in estimating nonmarket benefits is that most people do not have well formed values on the vast array of environmental resources and therefore resort to heuristics and simple protocols to construct dollar value estimates. These protocols, however, vary widely between individuals and many irrelevant issues, such as the appropriateness of the payment vehicle, who should pay, etc., often influence responses.[37] A few experiments suggest that stated willingness to pay for existence values may be two-to-ten times higher than actual contributions or payments.[38]

Despite the theoretical and methodological difficulties, quantifying the benefits of achieving clean water has often changed the character of the debate.[39] It is typical that the costs required to achieve water quality are highly concentrated but benefits are widely distributed. Because benefits to any one user may be quite small, only bearers of costs would have a great incentive to change the outcome of a debate. Given the public goods nature of many water quality improvements, a CV analysis may show that benefits, though widely dispersed, far outweigh costs.[40]

In sum, while certain benefits of water quality are reflected in markets and are relatively easy to measure, many are not. The nonmarket benefits of water quality are important and potentially very large. Measuring these benefits, however, presents unique challenges, which have been addressed by a number of ingenious nonmarket

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35. For this reason, CV questionnaires often attempt to convince respondents that their answers will have no effect on actual outcomes.

36. RIBAUDO & HELLERSTEIN, *supra* note 31, at 22-23.

37. David A. Schkade, *Issues in the Valuation of Environmental Resources: A Perspective from the Psychology of Decision Making*, *in* U.S. ARMY CORPS OF ENGINEERS, PUB. NO. IWR REPORT 95-R-2, REVIEW OF MONETARY AND NONMONETARY VALUATION OF ENVIRONMENTAL INVESTMENTS app. B (Timothy D. Feather et al. eds., 1995).

38. John B. Loomis, *Use of Non-market Valuation Studies in Water Resource Management Assessments*, 109 WATER RES. UPDATE 5, 6 (1997). However, estimated use values for hunting, fishing, rafting, camping, etc., employing CV willingness to pay results, were generally found to be slightly less than estimates based on actual behavior methods, *e.g.*, travel cost. *Id.*

39. *See generally id.*

40. *Id.*

techniques, CV being among the most promising. The accuracy, completeness, and potential bias of such techniques, especially CV, remain contentious issues. Despite these challenges, estimating benefits is an important element in the overall evaluation of the TMDL program and for UAAs and is being pursued by the EPA.[41]

Cost-Effectiveness Analysis

Cost-effectiveness analysis circumvents the difficulty and controversy of estimating economic benefits by focusing on the costs of achieving a quantified non-economic objective. Once this objective, *e.g.*, a water quality measure, has been chosen, costs of alternative strategies that reach the objective can be compared. Because estimating the cost of water quality measures is more reliable than estimating the expected benefits associated with those measures, cost-effectiveness analyses generally engender less controversy than cost-benefit analyses.

Cost-effectiveness analyses estimate the costs and environmental effectiveness of defined control measures or combinations of measures. In the TMDL context, effectiveness is often measured as a reduction in ambient pollutant load. Cost-effectiveness ratios are calculated by dividing the cost of a control measure by its effectiveness.[42] This ratio yields the dollar cost of achieving a one-unit improvement in the effectiveness measure. Cost-effectiveness ratios can easily be compared across control measures to determine the most cost-effective measures, *i.e.*, those with the lowest ratios. To achieve a water quality target in cases where any one control measure does not single-handedly achieve the target, the most cost-effective solution is typically found by applying the most cost-effective measure first, followed by the second most cost-effective measure, etc., until a specified water quality target is reached.[43]

Cost-effectiveness analysis is particularly well adapted to informing the decision making process for individual TMDLs for the following reasons: (1) numeric water quality targets are inherently part

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41. "While the estimation of benefits is traditionally difficult, EPA is working to develop improved models for describing benefits in both qualitative and quantitative terms." Proposed TMDL Rule, *supra* note 28, at 46,043.

42. Cost-effectiveness ratios can also be calculated by dividing effectiveness by cost, in which case, higher ratios would be preferable to lower ratios.

43. Cost-effectiveness ratio analysis, as outlined above, minimizes costs within the limitations imposed by ratio analysis, *e.g.*, linearity and additivity. Under more complex (and realistic) assumptions, cost-effectiveness analysis requires more completely defined relationships between costs, water quality indicators, and control measure adoption. Complex relationships among varying levels and combinations of control measures can often be addressed within a generalized cost minimization framework capable of solving for cost minimizing levels of control measures.

of the TMDL process;[44] (2) the particular means of attaining water quality targets are not prescribed, allowing flexibility and the comparison of alternative control measures; (3) descriptions of these control measures and their effectiveness are also required elements in TMDL implementation plans;[45] (4) considering implementation costs of alternative control is not prohibited and, in fact, is encouraged;[46] (5) from a welfare economic perspective, minimizing or reducing the cost of achieving water quality targets increases social welfare; and (6) compared to cost-benefit analyses, cost-effectiveness analyses are relatively straightforward and less controversial. Individual TMDL assessments, thus, are natural candidates for the application of cost-effectiveness analyses. Despite the suitability of cost-effectiveness analysis within the TMDL framework, rigorous analyses have been pursued for few TMDLs, and there remains a large potential for improving the economic performance of TMDL allocations.

EQUITY CONSIDERATIONS

Equity considerations often take on large proportions within TMDL deliberations because remedies are not prescribed but are the product of a human decision-making process. In many cases, entities belonging to different sectors, *e.g.*, industrial, agricultural, and urban, are jointly responsible for exceeding ambient standards. Reducing loads to TMDL targets can be accomplished by any combination of control measures, provided that total load reductions achieve the TMDL target. Cost minimization can be used as the sole criterion for assigning loads, but such assignments may be rejected because they are not deemed

44. Identification of pollutants and quantification of pollutant loads meeting water quality standards is one of ten minimum elements required for TMDL submissions to EPA. Proposed TMDL Rule, *supra* note 28 at 46,050; *see also* EPA, PUB. NO. EPA 841-D-99-001, DRAFT GUIDANCE FOR WATER QUALITY-BASED DECISIONS: THE TMDL PROCESS 3-3 (2d ed. 1999).

45. Proposed TMDL Rule, *supra* note 28, at 46,051; see also, EPA, *supra* note 44, at 3-22.

46. "TMDLs continue to provide for tradeoffs between alternative point and nonpoint source control options so that cost effectiveness, technical effectiveness, and the social and economic benefits of different allocations can be considered by decision-makers." Proposed TMDL Rule, *supra* note 28, at 46,030; moreover, EPA has characterized the TMDL process as a "cost-effective framework" for achieving water quality; Letter from Charles J. Fox, Asst. Administrator, EPA, to The Honorable Bud Shuster, Chairman of the House Transportation and Infrastructure Committee, U.S. House of Representatives (Apr. 5, 2000) (on file with author). EPA also explicitly and repeatedly refers to the legitimacy of employing cost-effectiveness in assessing control measures in its Draft Guidance, *e.g.*, "The TMDL process allows for alternative point and nonpoint source control strategies that provide decision makers with an opportunity to compare the cost-effectiveness and efficiency of different pollutant reduction activities or controls and the social and economic benefits of alternative allocation approaches." EPA, *supra* note 44, at 1-3.

equitable. The flexibility of the TMDL approach, thus, creates challenges to developing implementation plans that are deemed equitable to all parties, and, if allocations acceptable to all parties *are* found, these allocations may compromise economic efficiency.

Control costs are often concentrated in particular industrial sectors or entities. The fairness of how to assign these costs is almost always a contentious issue within TMDL deliberations. Subsidizing water quality improvements by transferring control costs from a small group of dischargers to broader publics is often viewed as an equitable means of distributing costs. State and federal funds can often be tapped to defray expenses associated with new control obligations.[47] Such transfers can promote the acceptance of new controls. When such funding is not available, the legality of mandating new control obligations, especially for nonpoint sources, is often tested.

Equity criteria can be analyzed by rigorous analytical methods, such as the attainment of a Pareto improvement.[48] In practice, however, equity issues are almost always dealt with in a subjective manner. Public participation forums within the TMDL process often provide official avenues through which equity issues are addressed.[49] Although load allocations are ultimately assigned and approved by state EPAs, most states assemble advisory boards of watershed stakeholders, typically composed of public servants, environmental interests, and industry representatives, to recommend key TMDL decisions, including load allocations. Recommended load assignments, thus, can be viewed as a product of internal political dynamics among advisory board members representing various interests. Within this context, appeals to equity and

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47. A prime example is the Natural Resources Conservation Service's Environmental Quality Incentive Program (EQIP), which provides up to a 75 percent cost share to agricultural producers for implementing practices that enhance environmental quality. *See* NAT. RESOURCES CONSERVATION SERV., ENVT'L QUALITY INCENTIVES PROGRAM FACT SHEET, June 2003, *at* http://www.nrcs.usda.gov/programs/farmbill/2002/pdf/EQIPFct.pdf (last visited Sept. 24, 2003).

48. Within a TMDL context, a Pareto improvement would necessitate transfers from the public (beneficiaries) to entities assigned control obligations such that all parties are at least as well off as before (*see supra* note 22).

49. The public participation process is an important aspect of the TMDL process and is strongly encouraged by EPA. *Supra* note 44, at 1-10 and 3-34–3-36.

> Historically, the EPA's policy has been that there should be full and meaningful public participation at the States, Territories and authorized Tribes level in both the listing and TMDL development processes. As such, EPA has encouraged States, Territories and authorized Tribes to carry out public participation consistent with their own public participation requirements.

Proposed TMDL Rule, *supra* note 28, at 46,038; the proposed regulation also requires that states provide the public with at least 30 days to review and comment on TMDLs, including allocations prior to submission to EPA. *Id.*; EPA, *supra* note 44, at 1-10.

fairness are perhaps the most potent and effective forms of persuasion influencing advisory board recommendations.

Any number of subjective criteria may be proposed to guide TMDL load allocations.[50] It may be deemed equitable for each identified source, or group of sources, to reduce their water quality impact by roughly the same percentage or absolute amount. Imposing pollution reduction obligations on small businesses and family farms, especially when economic viability is jeopardized, may be deemed unfair, as well as politically infeasible. While implementation strategies deemed equitable may significantly depart from cost-effective allocations, correctly structured incentive-based mechanisms potentially overcome this shortcoming.

POLICY INSTRUMENTS

Poor water quality is symptomatic of market failure, providing a rationale for collective intervention. This intervention can take the form of direct regulation, prohibition, or one of a variety of incentive-based mechanisms. Complete prohibition of discharge is sometimes resorted to in the case of extremely toxic substances, where even very small amounts might cause harm. Regulatory approaches typically prescribe specific control remedies and allow little flexibility in the means of achieving goals. By contrast, incentive-based regulation encourages behavior that reduces pollution through market forces and signals, allowing at least some degree of flexibility in the means of achieving goals. This flexibility provides opportunity for cost savings.

The NPDES permit system for point sources, which bases acceptable discharges on best available technology, is a prime example of a regulatory approach. Criticisms regarding the regulatory approach in general, and NPDES in particular, focus on issues of economic efficiency, *e.g.*, that it is overly prescriptive and lacks flexibility; that it seeks to regulate inputs rather than outputs (water quality); that it does not adequately consider costs; and even that it has reduced real incomes.[51] These criticisms help explain the recent rise to prominence of incentive-based regulatory approaches.

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50. In its Draft Guidance, EPA proposes several possible allocation methods: equal percent removal; equal concentrations; equal total mass per day, month, or year; equal reduction of raw load; equal ambient mean annual quality (mg/l); equal cost per mass of pollutant removed; percent removal proportional to raw load per day, month, year; most significant contributors achieve higher removal rates; seasonal limits based on cost-effectiveness analysis; and minimum total treatment cost. EPA, *supra* note 44, at 3-14

51. *E.g.*, Roger E. Meiners & Bruce Yandle, *Clean Water Legislation: Reauthorize or Repeal?*, in TAKING THE ENVIRONMENT SERIOUSLY (Roger E. Meiners & Bruce Yandle eds., 1993); Ackerman & Stewart, *supra* note 14.

Incentive-Based Mechanisms

Incentive-based mechanisms are advocated primarily as a means of reducing the financial burden of control costs while maintaining or making further progress on water quality. The defining hallmark of incentive-based systems is that they are designed to be incentive compatible, *i.e.*, by pursuing self-interests, environmental goals are simultaneously achieved. Unlike prescriptive regulation, incentive-based mechanisms can also promote the development and implementation of new pollution control technology.

Because of their theoretical efficiency characteristics, incentive-based mechanisms for achieving environmental quality have attracted the interest of economists for at least the past several decades. More recently, they have achieved a prominent place among the tools used by federal and state governments to address environmental problems.[52] Although historically shunned, incentive-based mechanisms have also gained the support of some environmental organizations.[53]

The theoretical appeal of incentive-based mechanisms is their potential to improve economic efficiency, *i.e.*, the same amount of pollution reduction can be obtained for a lower cost or a greater amount of pollution reduction can be attained at no greater cost. While generally viewed as a cost savings measure, making pollution control more affordable can also enhance its political acceptability and, hence, promote additional pollution control.[54] Incentive-based mechanisms for improving environmental quality include emission fees or taxes, tradable permits, deposit-refund systems, subsidies for pollution control, removal of subsidies with negative environmental impacts, reductions in market barriers, and performance standards.[55] The potential of the two most

52. EPA, PUB. NO. EPA-240-R-01-001 (2001), THE UNITED STATES EXPERIENCE WITH ECONOMIC INCENTIVES FOR PROTECTING THE ENVIRONMENT 1, *available at* http://yosemite.epa.gov/ee/epa/eermfile.nsf/vwAN/EE-0216B-01.pdf/$File/EE-0216B-01.pdf (last visited Sept. 25, 2003).

53. *See generally* Oates, *supra* note 4; examples from the environmental community include Paul Faeth, *Market based Incentives and Water Quality* (World Resources Institute, Oct. 1999), *at* http://www.wri.org/incentives/faeth.html (last visited Sept. 25, 2003); ENVTL. DEFENSE FUND, FROM OBSTACLE TO OPPORTUNITY: HOW ACID RAIN EMISSIONS TRADING IS DELIVERING CLEANER AIR (2000), *at* http://www.environmentaldefense.org/documents/645_SO2.pdf (last visited Sept. 25, 2003).

54. Tietenberg, for instance, maintains that "[w]ith the inclusion of a tradable permits program for sulfur in the [acid rain bill], the compliance cost was reduced sufficiently to make passage politically possible." Tom Tietenberg, *Tradable Permit Approaches to Pollution Control: Faustian Bargain or Paradise Regained?*, *in* PROPERTY RIGHTS, ECONOMICS, AND THE ENVIRONMENT 1-4 (Michael D. Kaplowitz ed., 1999).

55. *See generally* Robert W. Hahn, *The Impact of Economics on Environmental Policy*, 39 J. ENVTL. ECON. & MGMT. 375 (2000); Robert N. Stavins, *Market Based Environmental Policies,*

prominent incentive-based mechanisms (pollution taxes and tradable permits) to meet TMDL objectives are investigated below.

Pollution Taxes

Economists have long recognized the allocational inefficiencies associated with environmental externalities. Prior to World War II, the noted economist Arthur Cecil Pigou recognized that when there are divergences between private and social costs, self interest will not lead to maximum social welfare and consequently "certain specific acts of interference with normal economic processes may be expected, not to diminish, but to increase the dividend."[56] Pigou proposed an externality tax on pollution to internalize the damages caused by pollution. During his time, Pigou's pollution tax was regarded as an academic exercise and did not gain practical significance. The environmental consciousness of the 1970s, however, spurred a revival of interest in pollution taxes as a solution to environmental externalities, and several European countries and Japan adopted pollution taxes. The consensus among economists is that these fees have typically not had noticeable effects on pollution because they have not been set at levels that affect behavior.[57]

A simple economic model of water quality provision suggests that pollution taxes or tradable permits within a cap yield identical results if the cap or tax is appropriately set. This conclusion is illustrated by a simple graphic where water quality is represented on the horizontal axis, control cost is represented on the vertical axis, and an increasing marginal cost of water quality (a water quality supply function) is plotted within the graph (Figure 1).[58] A second horizontal axis depicts the inverse relationship between water quality and pollution.

(Resources for the Future, Discussion Paper 98-26, Mar. 1998), *available at* http://www.rff.org/rff/Documents/RFF-DP-98-(1998) (last visited Sept. 25, 2003); EPA, *supra* note 52.

56. A.C. PIGOU, THE ECONOMICS OF WELFARE 136 (1962) (1920).

57. *See, e.g.*, Hahn, *supra* note 55, at 379-80.

58. Baumol and Oates show that, under conditions of uncertainty in either marginal costs or marginal social damages of pollution control, either a pollution tax or a tradable permit program may yield higher expected social welfare, depending on the shapes of the marginal cost and marginal social damages curves. WILLIAM J. BAUMOL & WALLACE E. OATES, THE THEORY OF ENVIRONMENTAL POLICY 61-75 (2d ed. 1993).

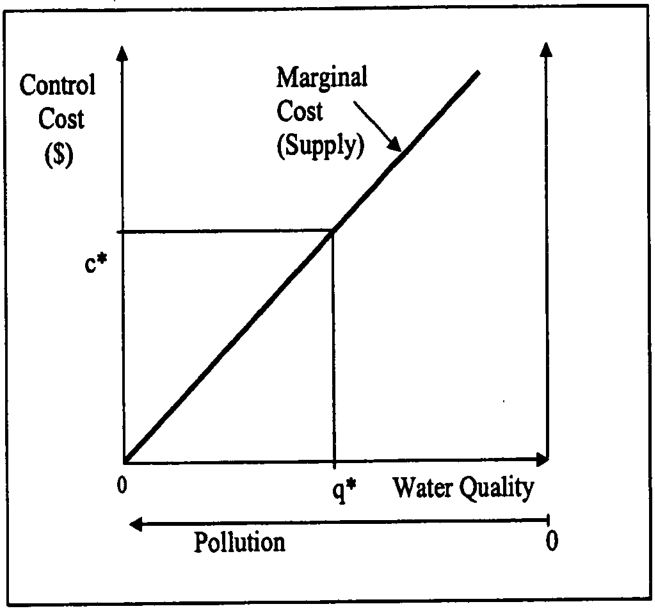

FIGURE 1: Theoretical Equivalence of a Pollution Tax (c*) and Pollution Cap (q*)

Any point on the marginal cost function can be mapped to a unique level of water quality and marginal cost; this is shown for one point where c* maps to q*. For this point, a pollution cap can be represented by vertical line at q* while the equivalent pollution tax could be represented by a horizontal line at c*, each intersecting the marginal cost function at the same point. For a given pollution tax (c*), the firm would have an incentive to reduce pollution as long as the pollution tax exceeded control costs, *i.e.*, up to the point q*. This level of water quality could also be achieved by directly specifying a pollution cap at q*, in which case cost minimizing behavior at the firm would dictate a marginal cost of pollution reduction at c*, or a level equal to the pollution tax. Despite the theoretical equivalence of pollution taxes and "cap and trade" programs, a number of practical advantages are often cited in favor of tradable permits, especially when attaining ambient standards is an overriding consideration, as is the case with the TMDL program.

Tradable Permits

The theoretical underpinnings of a tradable permits system have been traced to Coase,[59] who, in his classic work, *The Problem of Social Cost*, forcefully argued the importance of clear liability rules and bargaining between parties to achieve socially desirable levels of harmful effects.[60] Practical applications of Coase's basic premise were extended to air[61] and water pollution.[62] Whereas Coase envisioned the bargaining of social harms between the producer and recipient of those harms resulting in an efficient outcome regardless of who was responsible, Dales set out the parameters of the type of "cap and trade" program in widespread use today. The basic components of the program involve setting a cap on the total waste load, issuing emission permits, the aggregate of which equals the cap, and allowing the sale and purchase of permits among polluters. Marketable credits are formed when dischargers with low mitigation costs reduce loads below permitted levels. These credits can in turn be sold to entities with high mitigation costs such that both parties gain, and savings in meeting the total waste load are achieved.

Historical precedent for tradable permit systems in the United States and other countries is found in the allocation of natural resources, *e.g.*, water supply and fisheries.[63] Acceptance of the tradable permits in the United States for pollution control can be traced to the introduction of "pollution offsets" by the U.S. Congress in 1977, which opened the door to emissions trading in the air arena.[64] In the years since, tradable emission permit systems have been successfully implemented at the federal level for the lead phaseout program, to reduce ozone-depleting chemicals under the Montreal Protocol, and for the sulfur allowance program.[65] A number of programs have also been developed at the state level. Emissions trading for air pollution has generally been deemed quite successful. Trading of sulfur dioxide emissions, for instance, is estimated to have resulted in a 13 percent reduction in compliance costs while producing health related benefits of nearly $570 million in 1995.[66]

59. *See, e.g.*, Tom Tietenberg, *Introduction*, in EMISSIONS TRADING PROGRAMS, VOLUME II XV, XVI (Tom Tietenberg ed., 2001).

60. Ronald Coase, *The Problem of Social Cost*, 3 J.L. & ECON. 1, 8 (1960).

61. Thomas D. Crocker, *The Structuring of Atmospheric Pollution Control Systems*, *in* THE ECONOMICS OF AIR POLLUTION 61-86 (Harold Wolozin ed., 1966).

62. J.H. Dales, *Land, Water, and Ownership*, 1 CANADIAN J. ECON., 791, 791-804 (1968).

63. *See generally* Bonnie G. Colby, *Cap and Trade Policy Challenges: A Tale of Three Markets*, 76 LAND ECON. 638 (2000).

64. *See* Tietenberg, *supra* note 54, at 3.

65. *Id.* at 3-6.

66. DALLAS BURTRAW & ERIN MANSUR, RESOURCES FOR THE FUTURE, THE EFFECTS OF TRADING AND BANKING IN THE SO_2 ALLOWANCE MARKET 2, Discussion Paper No. 99-25

The use of tradable permits for *water quality* control is of recent origin. In 1992, no water effluent markets were reported to exist,[67] whereas several functioning trading programs were in operation by the late 1990s.[68]

The advantages of "cap and trade" programs relative to pollution taxes are, first and perhaps foremost, that a new pollution tax on industry may be politically unacceptable whereas the issuing of permits produces a potentially valuable property right.[69] Second, under conditions of uncertainty, a permit system would guarantee the attainment of the specified pollution level, whereas a large degree of guesswork would be involved in setting the appropriate tax level. Several adjustments to the tax might be required in finding the level that would produce the sought-after level of pollution control. Third, the amount of pollution reduction achieved by a given pollution tax is also unstable over time. Other things being equal, economic growth would generate increased emissions roughly proportional to increased production, as more pollution producing products and services were demanded and produced.[70] On the other hand, given expected advances in technology, levels of pollution might decrease if more cost-effective pollution technology were implemented over time.[71] In sum, while pollution taxes and tradable permits each have relative advantages,

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(1999), *available at* http://www.rff.org/Documents/RFF-DP-99-25.pdf (last visited Sept. 25, 2003).

67. Zach Willey, *Behind Schedule and Over Budget: The Case of Markets, Water, and Environment*, 15 HARV. J.L. & PUB. POL'Y 391, 394 (1992).

68. *See generally* ENVIRONOMICS, A SUMMARY OF U.S. EFFLUENT TRADING AND OFFSET PROJECTS (1999), *available at* http://www.epa.gov/owow/watershed/trading/traenvrn.pdf (last visited Sept. 25, 2003) (of the 37 trading and offset programs summarized by Environomics, eleven were well along in being implemented, with trades under way or completed; five had specific trading mechanisms approved and were very near implementation; six had completed the development and program approval process, but no specific trades had yet been identified; 12 were in various stages short of program approval, including study, discussion, planning and/or development,; one was exclusively a study; and two were inactive or discontinued); *see also* EPA, *supra* note 52.

69. *See generally* Tietenburg, *supra* note 54.

70. Let L^* be the initial pollutant load produced by a firm before reduction efforts, L be the load produced after reduction efforts, and C be the cost of load reduction. Assuming a linear marginal cost of load reduction, $C = a(L^* - L)$, and that initial load is proportional to production (X), $L^* = bX$, then $C = a(bX - L)$. Given a pollution load tax of t, the firm will seek to reduce loads to such a level that its own pollution reduction efforts, at the margin, equal the tax, thus $t = C = a(bX - L)$. Solving for the equilibrium load, $L = bX - t/a$. This expression shows that ending load (L) is more than proportional to production (X), and asymptotically proportional, for a given load tax (t). On the other hand, if a load cap (D) is imposed, then $C = abX - aD$, showing that load reduction costs are more than proportional to production and asymptotically proportional.

71. For the equations in the preceding footnote, cleaner production technology would be represented by a lower value for coefficient b, while reduced costs for removing load would be represented by a lower value for a. Both of these changes would reduce both costs and loads for the pollution tax scenario.

tradable permit systems are generally considered preferable when achieving quantitative pollution targets is the paramount consideration. Tradable effluent permits, thus, are conceptually well suited to achieving TMDL objectives.

Despite the flexibility of implementation measures inherent in TMDLs, TMDL allocations may not exhibit the greatest degree of cost-effectiveness. This can be attributed to the manner in which TMDL allocations are decided and, as previously discussed, equity issues. A tradable permit system, however, holds the promise of reallocating any initial set of load assignments such that a least cost (or lower cost) allocation is attained. Whether or not a tradable permit system, in practice, is more efficient than its absence depends on the level of transactions costs relative to the gains from trade. In general, a trading system is deemed advisable only if the benefits from trading exceed associated transactions costs. Transactions costs include the costs to industry of quantifying their mitigation costs; finding trading partners; initiating a trade; and bargaining as well as administrative costs, often borne by the public, of setting up and maintaining a market for pollution credits. If load allocations and implementation strategies are initially made in a relatively efficient manner, there is little room for additional savings.

Trading has the most potential where there are large differences between marginal pollution control costs and potentially large amounts of pollution credits that can be traded. For these reasons, some researchers see the greatest potential in trades between point and nonpoint sources, since the cost of nonpoint mitigation is often believed to be much less than that of point source mitigation.[72] Point/nonpoint source trading, however, entails an additional set of challenges including the monitoring and measurement of nonpoint pollution and uncertainty regarding the effectiveness of nonpoint controls.[73]

Effluent trading is most appropriate for water bodies where pollutants are well mixed. If pollutants are not well mixed, *e.g.*, a situation likely to occur in a stream or river, and several points in the water body are used to evaluate effectiveness, conditions to trading may need to be applied such that improvements in one area do not lead to unacceptable degradation in other areas. For this reason, it is likely that sale of pollution credits to downstream entities would be more likely

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72. *See generally* DAVID LETSON ET AL., POINT-NONPOINT SOURCE TRADING FOR MANAGING AGRICULTURAL POLLUTANT LOADINGS: PROSPECTS FOR COASTAL WATERSHEDS (U.S. DEP'T OF AGRIC., AGRIC. ECON. REP. NO. 674,1993); David Letson, *Point/Nonpoint Source Pollution Reduction Trading: An Interpretive Survey*, 32 NAT. RESOURCES J. 219 (1992).

73. *See generally* LETSON, *supra* note 72; Kurt Stephenson et al., *Watershed-based Effluent Trading: The Nonpoint Source Challenge*, 16 CONTEMP. ECON. POL'Y 412 (1998).

than upstream purchases because upstream purchases and subsequent emissions may subject more river miles to impairment, whereas downstream purchases may not.[74] Thus, the spatial attributes that often characterize water pollution pose challenges to the development of viable markets and trading programs and reflect the fact that pollution in water bodies is generally more localized and spatially isolated than for air sheds. The potential for viable water pollution trading programs depends on a number of site-specific factors, many of them dictated by nature. Under conducive circumstances, a properly structured trading program can be a potentially useful tool for achieving cost-effective water pollution control, either within or outside the TMDL process.

Having achieved notable successes in the air arena, economists and policy makers have investigated the potential of applying the tradable permit concept to water pollution control, and several programs have been developed.[75] In 1996, EPA issued a *Draft Framework for Watershed-Based Trading*, which was designed to promote, encourage, and facilitate trading wherever possible provided that equal or greater water pollution control can be attained for an equal or lower cost.[76] This was followed in January 2003 by a *Water Quality Trading Policy*.[77] Originally planned to be included in the impaired waters (TMDL) rule, EPA broadened the scope of the program to all waters and promulgated the policy in national guidance.[78] EPA's *Water Quality Trading Policy* was designed to provide guidance for state agencies in developing and implementing trading programs particularly for nutrients and sediments either within or outside of TMDL programs.[79] EPA believes there is substantial potential for reducing the cost of CWA compliance through trading, estimating that "flexible approaches to improving water quality could save $900 million annually compared to the least flexible approach."[80]

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74. Phosphorus, for instance, is a conservative element (*i.e.*, it does not volatilize). Since most surface waters are phosphorus limited, discharge into a stream would potentially impair all downstream river miles from the discharge point.

75. *See* ENVIRONOMICS, *supra* note 68; EPA, *supra* note 52.

76. EPA, PUB. NO. EPA 800-R-96-001, DRAFT FRAMEWORK FOR WATERSHED-BASED TRADING, ii, xiv (1996), *at* http://www.epa.gov/owow/watershed/framwork.pdf (last visited Sept. 27, 2003).

77. EPA, FINAL WATER QUALITY TRADING POLICY (2003), *at* http://www.epa.gov/owow/watershed/trading/finalpolicy2003.html (last visited Sept. 24, 2003).

78. Inside Washington Publishers, *EPA Will Expand Water Trading Program in Non-Binding Guidance*, 23 INSIDE E.P.A. WEEKLY REPORT 88 (Mar. 22, 2002).

79. *See generally* EPA, *supra* note 77.

80. EPA, *supra* note 77, at 2 (citing EPA, PUB NO. EPA 841-D-01-003, THE NATIONAL COSTS OF THE TOTAL MAXIMUM DAILY LOAD PROGRAM (DRAFT REPORT) (2001)).

A controversial issue in EPA's *Water Quality Trading Policy* is the inclusion of nonpoint sources in addition to point sources.[81] EPA does not have formal authority to regulate nonpoint source dischargers under the CWA, nor can most states regulate nonpoint sources under state statutes, instead relying on voluntary or incentive-based mechanisms.[82] EPA indicates that TMDL point source waste load allocations and nonpoint source load allocations should be used as a baseline for the generation of pollution reduction credits.[83] Nonpoint sources, thus, might produce and sell pollution reduction credits in exchange for adopting land management practices that reduce expected loads below the TMDL baseline. Because nonpoint sources, however, are generally not prohibited from engaging in practices that might increase nonpoint loads, the purchase of credits to engage in such practices would generally not be required. At a minimum, it would be very challenging, both politically and legally, to require that another entity reduce loads based on a nonpoint source's unilateral decision to adopt load-increasing practices.[84]

A second problem in allowing trades between point and nonpoint sources involves uncertainties in measuring nonpoint loads; uncertainties in determining, a priori, what load reductions will be for nonpoint best management practices under a given set of conditions; and uncertainties in the timing of nonpoint source loads, which are weather dependent. EPA has suggested a number of approaches to compensate for nonpoint source uncertainty. These include monitoring to verify load reductions, using a greater than 1:1 trading ratio between nonpoint and point sources, using conservative assumptions, retiring a percentage of nonpoint source reductions for each transaction, and establishing a reserve pool of credits to compensate for unanticipated shortfalls.[85]

Finally, biophysical analyses of nutrient loads indicate that gross loads are often poor measures of impairment because significant differences in the timing and character of point source versus nonpoint source loads render them fundamentally non-equivalent from an impairment standpoint.[86]

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81. *Id.* at 9.

82. The CWA expressly excludes agricultural storm water discharges from point source status. Federal Water Pollution Control Act § 502(14), 33 U.S.C. § 1362 (2000). Most states with delegated authority to implement the NPDES program have also adopted this stance.

83. EPA, *supra* note 77, at 5.

84. This nonpoint source "loophole" is not only a potential pitfall for water quality trading programs, but for the TMDL program, generally.

85. EPA, *supra* note 77, at 9.

86. For instance, stream impairment may critically depend on the timing of pollutant discharges. A small nutrient load discharged on a continuous basis, *e.g.*, from a wastewater treatment plant, generally produces greater impairment than a much larger load

Despite recent efforts by EPA and some states to encourage the implementation of watershed trading, programs have been developed at relatively few locations and trading within these programs has been limited, such that few examples of successful trades can be found. It remains to be seen whether vigorous implementation of the TMDL program, alongside official federal and state endorsement and encouragement of effluent trading, will greatly increase the use of trading as suggested by EPA,[87] or if spatial and administrative challenges will continue to limit its adoption to a very small percentage of impaired or potentially impaired waters.

CASE STUDY: NORTH BOSQUE RIVER TMDLS

The following case study provides an example of an economic analysis that was performed in conjunction with an actual TMDL. Formal economic investigations within the context of TMDL decision making are, at present, rare. In cases where impairment is attributable to several sources or several viable control measures are available, however, a cost-effectiveness analysis of control measures may potentially identify significant cost savings in meeting TMDL targets.

Background

The North Bosque River, located in north central Texas, flows through six small communities ranging in population from 500 to 16,000, before draining into Lake Waco, a drinking water source for approximately 150,000 people located in and around the City of Waco. The headwater area of the North Bosque River basin is located in the state's top dairy production region. An estimated 43,000 dairy cows at approximately 106 dairies are located within the watershed. Although comprising only three percent of the land area, dairy waste application fields (WAFs) contribute an estimated 35 percent of soluble phosphorus

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discharged over a short period of time, *e.g.*, storm runoff of nutrients from agricultural land. *See, e.g.*, Keith Keplinger & Ron Jones, *Socio-Economic and Biophysical Challenges to Achieving Clean Water Through TMDLs: Two Texas Examples*, 1 WATER RESOURCES IMPACT, Nov. 1999, at 15. Fred Lee writes, "It is extremely important that pollutant-trading programs be based on available forms of nutrients and not on total nutrients, which include large amounts of unavailable nutrients. Numerous technically invalid pollutant-trading programs have been developed, which incorporate trades of discharge of unavailable nutrients for discharge of available nutrients." G. Fred Lee, *Evaluating Nitrogen and Phosphorus Control in Nutrient TMDLs*, 3 STORMWATER (Jan./Feb. 2002), *available at* http://www.forester.net/sw_0201_evaluating.html. It is possible that impairment units other than loads, *e.g.*, time-weighted concentration units, could be traded, although there is little evidence that such scenarios have been explored.

87. EPA, *supra* note 52, at 99-100.

loadings to the watershed.[88] Wastewater treatment plants (WWTPs) in the communities along the North Bosque River also provide a significant portion of watershed phosphorus loads and contribute disproportionately to instream concentrations. Elevated levels of soluble phosphorus in lakes and streams can lead to accelerated eutrophication, a condition characterized by excessive algal growth and associated effects such as fish kills and unpleasant taste and odor. Sporadic algal blooms and associated taste and odor problems in drinking water from the lake have long concerned the City of Waco and have focused attention on the North Bosque River as a possible source of impairment.

Since 1992, when the Texas Commission on Environmental Quality (TCEQ)[89] first compiled biannual 303(d) lists, the North Bosque River has been included on the state's impaired waters list. In the 1998 list as well as former lists, the TCEQ found the river impaired under narrative water quality criteria related to excessive aquatic plant growth.[90] Instream bioassays conducted in 1998 provided strong evidence that phosphorus was the limiting nutrient for algal growth under most conditions.[91] Loading studies revealed that WAFs and municipal WWTPs were the major controllable sources of phosphorus.[92] Additional research indicated that soluble phosphorus, measured as soluble reactive phosphorus (SRP), was statistically better correlated to algal levels than total phosphorus.[93]

The Bosque River Advisory Committee (BRAC) was formed to comply with Texas's TMDL public participation requirements, while a technical work group consisting of university and public agency scientists was assembled to provide technical support to the BRAC. The full BRAC or its subcommittees met a total of 20 times between February 1998 and August 2000 but did not reach consensus and, consequently,

88. ANNE MCFARLAND & LARRY HAUCK, TEX. INST. FOR APPLIED ENVTL. RESEARCH, PUB. NO. PR 99-11, EXISTING NUTRIENT SOURCES AND CONTRIBUTIONS TO THE BOSQUE RIVER WATERSHED, i-ii (1999).

89. Effective September 1, 2002, the former Texas Natural Resource Conservation Commission (TNRCC) formally changed its name to the Texas Commission on Environmental Quality.

90. TEX. NAT. RESOURCES CONSERVATION COMM'N, TWO TOTAL MAXIMUM DAILY LOADS FOR PHOSPHORUS IN THE NORTH BOSQUE RIVER 2 (2001). The North Bosque River is also listed on Texas's 303(d) lists for impairments other than excessive aquatic plant growth; the TMDL, however, is only for phosphorus.

91. Marty Matlock et al., *Development and Application of a Lotic Ecosystem Trophic Status Index*, 42 TRANSACTIONS AM. SOC'Y. AGRIC. ENG'RS. 651, 651-54 (1999).

92. MCFARLAND & HAUCK, *supra* note 88, at i.

93. RICHARD L. KIESLING ET AL., TEX. INST. FOR APPLIED ENVTL. RESEARCH, PUB. NO. TR0107, NUTRIENT TARGETS FOR LAKE WACO AND NORTH BOSQUE RIVER: DEVELOPING ECOSYSTEM RESTORATION CRITERIA 3-4 (2001).

provided no recommendations to the TCEQ on key TMDL targets or allocations.

In February 2001, the TCEQ promulgated TMDLs for phosphorus in the North Bosque River,[94] setting as its endpoint a significant reduction in SRP average annual loading and annual average concentrations at various sites. The numeric statement of this goal was to reduce average total-annual loading of SRP by approximately 50 percent for the entire North Bosque River watershed. Numeric reductions in SRP loads and concentrations, thought to be achievable based on computer simulations, were quantified in the TMDLs. Because of the TMDLs' emphasis on reducing SRP concentrations as well as loads, the economic analysis considered reductions in both SRP loads and concentrations as equally important dual goals.[95]

Methodology

A series of computer simulations were conducted using the Soil and Water Assessment Tool (SWAT)[96] to simulate baseline conditions and the implementation of several control measures designed to reduce SRP loads and concentrations. Baseline and P control scenarios were simulated assuming effluent discharges from WWTPs and cow numbers at permitted levels.[97] Implementation costs at the watershed level for P control scenarios were also estimated based on fully permitted levels.

The baseline scenario simulated dairy waste application at the nitrogen rate and nutrient concentrations in WWTP effluent at current

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94. TEX. NAT. RESOURCES CONSERVATION COMM'N, *supra* note 90, at 2.

95. Phosphorus loadings from WAFs and WWTPs vary markedly in character and timing. Large rainfall events cause runoff from WAFs, producing large but infrequent phosphorus (P) loads to the river. By contrast, annual P loads for WWTPs are small compared to those of WAFs but contribute disproportionately to instream time-weighted concentrations, due to the continuous release of effluent.

96. SWAT, developed by the USDA Agricultural Research Service, is a continuous-time, long-term, simulation model designed to predict the impact of management on water, sediment, and agricultural chemical yields in large ungauged basins. *See generally* Jeffery G. Arnold et al., *Large Area Hydrologic Modeling and Assessment—Part 1: Model Development*, 34 J. AM. WATER RESOURCES ASS'N 73 (1998); model documentation is provided in SUSAN L. NEITSCH ET AL., USDA AGRIC. RESEARCH SERV., GRASSLAND SOIL AND WATER RESEARCH LAB., SOIL AND WATER ASSESSMENT TOOL USER'S MANUAL, VERSION 2000 (2002), *available at* http://ftp.brc.tamus.edu/pub/swat/doc/swatuserman.pdf (last visited Sept. 25, 2003).

97. Simulating fully permitted levels may satisfy future growth requirement for TMDLs, since fully permitted cow numbers and WWTPs flows are not anticipated for many years. As of year 2000, the aggregate annual discharge at the North Bosque WWTPs was 57 percent of permitted levels while actual dairy cow numbers in the watershed were 63 percent of permitted levels.

median values.[98] Three dairy Best Management Practices (BMPs) were simulated: (1) haul out of all solid dairy manure (haul out), (2) application of manure at the crop P requirement rate (P rate),[99] and (3) a reduction of dietary P in animal feed (P diet).[100] A reduction of total P in the effluent of six WWTPs discharging into the North Bosque River to one mg/l and two combined strategies were also simulated. Combined strategy I simulated simultaneous adoption of the P rate, P diet, and P reduction at WWTPs. In addition to these measures, combined strategy II assumed that WAFs were limited to permitted areas and that excess manure would be hauled out of the watershed.[101] Santhi et al. provide additional methodological details.[102]

Three sites on the North Bosque River were analyzed: Stephenville, above Meridian, and Valley Mills, which were chosen to represent the headwater area, a site approximately midway down the watershed, and the most downstream site, respectively. Simulations for existing conditions, reported in the North Bosque River TMDLs, indicated that loads must be reduced by an average of 50 percent and concentrations by an average of 49 percent from current levels in order to reach the numeric targets.[103] To achieve the same numeric targets from the fully permitted levels, however, SRP loads must be reduced by an average of 65 percent and SRP concentrations by an average of 70 percent.

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98. Grab samples at WWTP outfalls between 1993 and 2002 indicate median total P concentrations in effluent ranging between 2.2 mg/l and 3.7 mg/l for the six Bosque WWTPs. KEITH KEPLINGER ET AL., TEX. INST. FOR APPLIED ENVTL. RESEARCH, PUBL'N. NO. TR0312 (2003), ECONOMIC AND ENVIRONMENTAL IMPLICATIONS OF PHOSPHORUS CONTROL AT NORTH BOSQUE RIVER WASTEWATER TREATMENT PLANTS 13, *available at* http://tiaer6.tarleton.edu/pdf/TR0312.pdf (last visited Sept. 25, 2003).

99. Published P rates are designed to supply crops with amounts of P equal to plant uptake requirements plus unavoidable losses to soil and water. P rates for manure application are typically several times less than nitrogen (N) based rates because the N:P ratio for crop requirements is several times higher than the N:P ratio of most manures.

100. The P diet BMP assumes a reduction of total P in lactating cow diets from 0.53 percent to 0.40 percent of dry matter, which is considered to more than meet phosphorus requirements for dairy cattle in the region. Dietary phosphorus in excess of requirements has been shown to have no beneficial effect on animal health or production.

101. This additional assumption requires that 59 percent of manure is hauled out of the watershed, since existing WAFs can accommodate only 41 percent of manure when applied at the P rate.

102. C. SANTHI ET AL., TEX. INST. FOR APPLIED ENVTL. RESEARCH, BLACKLAND RESEARCH & EXTENSION CTR., TEXAS AGRICULTURAL EXPERIMENT STATION, BRC REPORT NO. 01-34, USDA LAKE WACO/BOSQUE RIVER INITIATIVE: WATER QUALITY MODELING OF BOSQUE RIVER WATERSHED USING SWAT FOR THE ASSESSMENT OF PHOSPHORUS CONTROL STRATEGIES (2000).

103. Average percent reductions reported in the North Bosque TMDLs and in this report are the unweighted average of percent reductions needed at each site to achieve numeric SRP load and concentration targets. SRP concentrations are average annual time-weighted concentrations, *i.e.*, means of average annual daily concentrations. *See* TEX. NAT. RESOURCES CONSERVATION COMM'N, *supra* note 90, at 3.

Control measure effectiveness was calculated in terms of progress toward numeric targets and reported as a percentage in order to shift focus from absolute magnitudes of SRP loads and concentrations, which vary by orders of magnitude among the three sites, to progress toward TMDL numeric targets.[104] Effectiveness measures for the three sites were averaged (unweighted) to arrive at reported composite effectiveness measures for each control scenario.

Costs of the haul out and P rate dairy BMPs were estimated using the Farm Economic Model.[105] The economic benefits of reducing dietary P in cow diets were based on the prices and quantity changes of dicalcium phosphate and crushed limestone such that the existing calcium and reduced phosphorus requirements were met. The cost of implementing P reduction technology at six WWTPs located on the North Bosque River, such that total ambient P concentrations in effluent would meet a one mg/l limit, was estimated based on a site-specific engineering study.[106] Cost-effectiveness ratios for control scenarios were calculated by dividing control measure costs by scenario effectiveness. Cost-effectiveness ratios, thus, represent the cost of attaining a one percent improvement in effectiveness for P loads and concentrations.

Results

For each control measure and the combined strategies, Table 1 reports estimated annualized cost, environmental effectiveness, and cost-effectiveness ratios. Costs of the combined strategies are less than a summation of the costs of their components because savings are realized when the P rate BMP is combined with the reduced P diet. Haul off is the most expensive scenario considered, followed by P rate and P reduction at WWTPs. The P diet BMP produces financial benefits because dairy producers are able to reduce or eliminate the use of supplemental dietary P, which is typically the second most expensive component in lactating cow diets.

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104. For example, if an SRP concentration for the baseline scenario is simulated at 80 parts per billion (ppb), the numeric target is 30 ppb, and a control scenario reduces the SRP concentration from 80 to 60 ppb, then the control measure reduces SRP concentrations by 20 ppb while a reduction of 50 ppb is needed to achieve the numeric target. Hence, the effectiveness measure for the control strategy (progress toward the target) is reported as 20/50 or 40 percent.

105. The Farm Economic Model is described in EDWARD OSEI ET AL., TEX. INST. FOR APPLIED ENVTL. RESEARCH, PUB. NO. PR0002, LIVESTOCK AND THE ENVIRONMENT: A NATIONAL PILOT PROJECT; ECONOMIC AND ENVIRONMENTAL MODELING USING CEEOT 21 (2000), *available at* http://tiaer6.tarleton.edu/pdf/PR0002.pdf (last visited Sept. 25, 2003).

106. CAMP DRESSER & MCKEE, INC., NORTH BOSQUE RIVER PHOSPHORUS REMOVAL STUDY FOR SIX WASTEWATER TREATMENT PLANTS (2001).

Table 1. Annual Cost, Environmental Effectiveness, & Cost-effectiveness Ratios of Phosphorus Control Measures, North Bosque River

	Dairy BMPs			WWTP	Combined Strategies	
	Haul Off	P Rate	P Diet	1 mg/l TP	I[a]	II[b]
Annualized Cost ($1,000)	4,656	2,964	(1,252)	534	2,574	4,117
Environmental Effectiveness						
Loads (% reduction)	68	41	21	29	72	87
Concentrations (% reduction)	12	6	3	77	84	87
Cost-Effectiveness Ratios						
Loads ($1000/% Effectiveness)	69	73	(59)	18	36	47
Conc. ($1000/% Effectiveness)	399	511	(361)	7	31	48

[a] Combined strategy I simulates simultaneous adoption of the P rate, P diet, and P reduction at WWTPs. [b] In addition to the control actions in combined strategy I, combined strategy II simulates that waste application fields are limited to permitted areas and that excess manure is hauled out of the watershed.

Focusing on the environmental effectiveness of individual control measures, the study concluded that the haul out scenario is the most effective measure for reducing SRP loads (reducing loads by 68 percent), followed by P rate (-41 percent), P reduction at WWTPs (-29 percent), and P diet (-21 percent). The relative effectiveness of control measures in reducing SRP concentrations, however, is quite different. Results indicate that P reduction at WWTPs is by far the most effective control measure for reducing SRP concentrations (reducing concentrations by 77 percent), followed by haul out (-12 percent), P rate (-6 percent), and P diet (-3 percent). P reduction at WWTPs is simulated to be more than six times as effective in reducing SRP concentrations as the haul out scenario. The haul out scenario, however, is more than twice as effective in reducing SRP loads than P reduction at WWTPs.

Theory indicates that the cost of achieving environmental targets can be minimized by applying the most cost-effective measures first, followed by less cost-effective measures, until targets are reached. Following this procedure, the reduced P diet would be the first candidate for inclusion into a TMDL implementation strategy because it saves producers money while reducing SRP loads and concentrations: a classic win-win scenario. P reduction at WWTPs would be the next candidate for consideration because its cost-effectiveness ratios for both SRP loads and concentrations are lower than those of the remaining control measures. The simultaneous adoption of these two control measures, however, would not come close to achieving either the SRP load or concentration targets.[107]

Effectiveness values reported in Table 1 indicate that the combined strategies come closer to meeting numeric SRP load and concentration targets than any individual measure but still do not reach targets, which would be indicated by a value of 100. Combined strategy II, the most intensive of the scenarios considered, comes closest to meeting targets. Simulation results indicate that combined strategy II achieves 87 percent of both SRP load and concentration reductions needed to achieve numeric targets. As previously indicated, the baseline for this analysis assumes dairy cow numbers and discharges at WWTPs at permitted levels. If there were little or no future growth within the time frame of the TMDL, then adoption of combined scenario II would likely come close to or meet numeric targets.[108] Thus, an implementation strategy approximating that of combined strategy II would appear to be needed to approach or possibly reach the numeric statement of the TMDLs' end points.

Discussion

In the presence of alternatives for achieving TMDL targets, costs can be reduced by implementing more cost-effective control strategies. This analysis suggests that the ability to substantially reduce costs for the North Bosque TMDLs is very limited because (1) both SRP loads and concentrations are targeted and (2) the levels at which those targets are set. Results indicate that all of the control measures considered would need to be implemented in order to approach or reach numeric TMDL targets for SRP loads and concentrations. These results are a reminder that the application of cost-effectiveness analysis to TMDL control alternatives may or may not reveal more efficient TMDL allocations. Nonetheless, a more complete understanding of control measure costs relative to their effectiveness provided useful input into the North Bosque TMDL deliberations.[109]

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107. A scenario representing the combined implementation of the P diet in combination with P reduction at WWTPs was not simulated, but effectiveness would necessarily be less than that of combined strategy I.

108. Simulations using current conditions for the baseline were analyzed for TMDL committee deliberations, but a consistent and complete set of "current condition" simulations was not achieved and therefore is not presented.

109. This is not to say that the economic efficiency of proposed control strategies played a dominant role in discussions or in final TMDL allocations. Assessing the goals of individual TMDL stakeholder participants, TMDL committee dynamics, and agency behavior is beyond the scope of this article.

Interestingly, a post-TMDL analysis of phosphorus control focusing just on North Bosque River WWTPs revealed that up to $152,000 could be saved annually by allowing WWTPs to trade phosphorus credits, rather than mandating that each WWTP reduce its load by the same percentage.[110] Thus, while the adoption of all control measures considered appears necessary to achieve TMDL goals, considerable reduction in the costs to WWTPs could be achieved through the implementation of a phosphorus "cap and trade" program for WWTPs, assuming that all WWTPs were initially required to meet 1 mg/l effluent limits for P. Alternatively, more efficient initial load allocations among the six North Bosque WWTPs could also achieve an equivalent cost reduction. While the theoretical post trading load distribution among the six WWTPs would be the same as a most efficient initial load allocation (the largest WWTP would make all the reductions), the smaller WWTPS would not need to buy P credits and, hence, would incur no financial obligations under an efficient initial load allocation, the entire cost falling to the largest WWTP.

CONCLUSIONS

The shift of water quality policy from ambient standards to an effluent-based program with the passage of the 1972 CWA was consistent with economic rationality. To wit, pre CWA ambient-based water quality legislation proved infeasible and effete, despite its theoretical appeal, while, for all its flaws, the effluent-based NPDES program achieved limited success. While criticized for its theoretical inefficiency and reliance on prescriptive measures, the efficacy of the NPDES program lay in its simplicity combined with strong federal enforcement. Resorting to TMDLs, where effluent standards alone are inadequate, can also be viewed as economically rational, especially since nonpoint source pollution is not covered by NPDES. Three reasons can be provided for the premise that the TMDL program is more appropriate, economic, and will be more effective than its ambient-based predecessors: (1) TMDLs are required only where existing water quality programs have proved inadequate in achieving ambient standards; (2) advances in science, technology, and environmental economics have made the ambient-based TMDL approach eminently more feasible today

110. KEPLINGER ET AL., *supra* note 98, at iii. The analysis of the WWTP control measures presented in Table 1 assumes that all WWTPs would reduce loads to meet 1 mg/l effluent levels, while Keplinger et al., *id.*, allow the trading of phosphorus credits within a cap such that total emissions by all six North Bosque River WWTPs collectively do not exceed the cap, which is equal to the load that would be produced by all WWTPs reducing loads to meet the 1 mg/l limit.

than 30 years ago; and, finally, (3) the TMDL program includes substantial federal involvement, which was largely absent in earlier ambient-based programs.

Still, the administration and implementation of TMDLs is time consuming and expensive, and the supporting science often does not resolve a great deal of uncertainty in outcomes even with today's most advanced assessment tools. Estimating the benefits as well as the costs of the TMDL program is being pursued by EPA. Neither the estimation of costs nor benefits is required for *individual* TMDLs, since targets are based on ambient standards, not economic benefits. A committee formed by the National Research Council, however, recommends that both the costs and benefits of various levels of water quality be estimated and evaluated within UAAs, the procedure that informs the assignment of designated uses for water bodies.[111]

Within TMDL deliberations, estimating the costs of control measures is not only allowed but also encouraged. Numeric water quality targets are inherently part of the TMDL process and descriptions of control measures and their effectiveness are required elements in TMDL implementation plans. For these and other reasons, TMDLs are natural candidates for the application of cost-effectiveness analysis. Even if cost-effectiveness analyses are pursued, however, equity considerations or political dynamics may cause recommended or final TMDL allocations to significantly depart from cost-effective allocations. Properly structured incentive-based mechanisms, especially marketable permits, may potentially readjust allocations such that standards can be met at lower cost. Several "cap and trade" programs for water quality have been instituted within the last decade with mixed results. The EPA recently issued a water quality trading policy to encourage and provide guidance to states in developing and implementing trading programs for nutrient and sediment loads, and several states have initiated programs. This official endorsement will likely increase the implementation of trading programs, although their adoption will be far from universal. High transactions costs and the dynamic and spatial characteristics of water quality will limit the potential of successful trading programs to that portion of water bodies with amenable characteristics.

The North Bosque River TMDLs provide a case study of a formal economic analysis pursued within a TMDL. A cost-effectiveness analysis indicated that some of the proposed control measures were much more effective and cost-effective in reducing phosphorus loads while others were much more effective and cost-effective in reducing concentrations. Because reductions of both loads and concentrations were considered

111. *See* NAT'L RESEARCH COUNCIL, *supra* note 6, at 92.

equally important dual goals, the analysis indicated that all proposed control measures would be needed to approach or meet numeric TMDL targets. A post-TMDL analysis, however, showed that substantial savings to North Bosque River WWTPs could accrue if the trading of phosphorus credits were allowed to meet phosphorous control responsibilities. Alternatively, a more efficient initial allocation of control responsibility could achieve the same savings, although the entire cost of phosphorous control would be shifted to the largest WWTP in this case.

Costs and economic efficiency are important considerations for any policy decision, including individual TMDL allocations and the TMDL program itself. Although economic considerations were once deemed relatively unimportant (or even counter-productive) in the implementation of CWA policy, the flexibility of the TMDL program and its rise to prominence, in conjunction with a well developed environmental economics field, offers an amenable environment for conducting cost-effective analyses within individual TMDLs, possibly uncovering efficiencies. Where conditions are conducive, the trading of marketable pollution credits may further reduce costs. There is, therefore, great but heretofore underutilized opportunity to conduct economic analyses directed specifically at individual TMDLs or as part of TMDL assessments. Revealing control costs and economic efficiencies for control alternatives does not guarantee ultimate adoption of the most efficient control strategies, but it does bring critically important information to bear on TMDL decision making.