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Analysis of Water Quality at Watershed-Scale and Equity Considerations in Pollution Control and Management

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ABSTRACT: Proper management of water quality at watershed scale, especially due to non-point source pollutants, is of great importance where considerable revenue is generated to the local and national economies. Management of watersheds involves two important steps; first the accurate analysis of physical and other processes affecting flow, and fate and transport of pollutants of concerns, and second, the development of appropriate best management practices that can satisfy both environmental and stakeholder constraints. Significant work has been performed in the analysis of non-point source pollutants at watershed scale. Of the different methods, export coefficient approach has been popular in predicting the non-point source loadings to surface water pathways due to its simplicity and small data requirements. However, this approach has significant limitations because the method does not address hydrologic variability commonly present in all watersheds.

In the control and management of non-point source pollutants, most best-management practices are developed with the focus of cost minimizations or cost efficiency with little regard to stake holder concerns especially in equity. Due to this limitation, implementation of proposed best-management practices to most large watersheds with multiple stakeholders and land use activities fail because cost equity is not considered in the development stages.

In this work, we propose to address both these limitations in the analysis and management aspects of water quality. In the first step, we propose to modify the export coefficient method to include hydrologic variability and show that the results from the proposed approach have significantly better and reliable results over the classical export coefficient method. We further demonstrate the importance of considering both uncertainty and variability in the methodology using a Bayesian framework. In the second step, we will consider the tradeoffs between cost efficiency and cost equity in allocating pollution control responsibilities such that the framework can be adopted by decision-makers in a flexible manner. We show that cost efficiency and equity are important considerations in the management and the proposed methodology allows the decision-makers to develop a comprised solution based on environmental and stakeholder constraints.

INTRODUCTION

P-loading is modeling using export coefficients, which calculate annual pollutant loads to water bodies from a catchment as the sum of individual loads exported from each land use type as,

$$L = \sum_{i=1}^{n} E_i \times A_i \times I_i \qquad \dots (1)$$

The traditional P-export coefficient model defines the annual transport of nutrients from land uses to the watershed outlet without the representation of processes controlling this transport. This formulation of export coefficients is sensitive to changes in the land use area and P application rates, but not to changes in processes controlling the physical transport of total P, namely, run-off and soil erosion. Several studies have attempted to introduce process representation into the basic export coefficient model. These include weighing export coefficient values that are within a threshold distance of rivers (Johnes and Heathwaite, 1997; Johnes and O'Sullivan, 1989). Worrall and Burt (1999) extended export coefficients model to include the effect of soil reservoirs of nutrients. Endreny and Wood (2003) used watershed topographic index and buffer index to weight export coefficients values to represent wetting up, run-off, and subsequent capture likelihood.

EROSION-SCALED EXPORT COEFFICIENTS

An alternative formulation for the total P-export coefficient method that considers the transport process can be described as,

$$L = \Phi(R) \times \sum_{i=j}^{n} K_j \times A_j \times I_j \qquad \dots (2)$$

where R is the annual run-off (m^3); $\Phi(R)$ is the annual sediment discharge as a function of annual run-off (Kg); and K_j is the erosion-scaled export coefficient for land use j (Kg^{-1}).

Using this formulation, the annual export of P is a linear function of the total sediment discharge, which is a function of the volume of annual run-off and watershed characteristics. Thus, hydrologic controls on P transport are now present in the export coefficient approach. It should be noted that this formulation assumes negligible contribution of subsurface flow to P transport, which might not be true in some watersheds and has to be taken into account. Comparing Equations (1) and (2), the erosion-scaled export coefficient (K) is related to the original export coefficient (E) as,

$$K_j = \frac{E_j}{\Phi(R)} \qquad \dots (3)$$

The erosion-scaled export coefficient (K) translates the P application rate to an equivalent soil-P concentration. Hence, $\sum_{j=1}^{n} K_j \times A_j \times I_j$ represents an area weighted equivalent soil-P concentration, which when multiplied by the amount of sediment discharge will give total P exported from the watershed. Equivalent soil-P concentration is actually the P-sediment ratio in stream run-off. It should also be noted that the P-sediment ratio is typically much higher than a soil-P test as enrichment of P takes place during surface run-off (Sharpley *et al.*, 1999).

EXPORT COEFFICIENT UNCERTAINTY

Uncertainty in the export coefficient approach is inherently high because of its empirical nature that does not address explicitly the processes controlling P release and transport and the spatial heterogeneity within each land use class. Equation (2) addresses an important source of uncertainty in export coefficients, which is hydrologic variability. Another source of uncertainty is the difficulty of allocating exported nutrients to different land uses in a watershed. Because of this uncertainty, the ability to manage nutrient exports suffers since the proportion of nutrients supplied by each source is not known with sufficient certainty. Developing a realistic estimation of uncertainties in export coefficients is necessary to evaluate the confidence limits of model predictions, which can have a great impact on watershed management.

To estimate export coefficients uncertainty, Khadam and Kaluarachchi [2006a] proposed a generic framework for parameter uncertainty widely used in hydrologic modeling based on Bayes theorem. The Bayesian parameter estimation starts by defining the prior probability distribution for model parameters, $p(\theta)$, which is the initial belief about the parameter values from experience with similar models. The prior probability distribution of parameters, $p(\theta)$, is updated using the observed data, D, to develop a posterior probability distribution, $p(\theta|D)$. Bayes' theorem describes the updating process as,

$$p(\theta|D) = \frac{p(D|\theta) \cdot p(\theta)}{\int p(D|\theta) \cdot p(\theta) \cdot d\theta} \qquad \dots (4)$$

where $p(D|\theta)$ is the conditional probability or likelihood of observing D assuming that the value of parameters is θ. For n observed data points, the likelihood function is defined in a multiplicative form as,

$$p(D|\theta) = \prod_{t=1}^{n} p(D_t \mid \theta) \qquad \dots (5)$$

where $p(D_t \mid \theta)$ is defined assuming errors are white noise, to avoid explicit definition of the sampling distribution.

The implementation of the Bayesian parameter estimation involves sampling from a probability distribution, which can be achieved through Monte Carlo simulations. In this study, the Metropolis algorithm, which belongs to a class of the Monte Carlo method and based on Markov chains (MCMC), is used (Kuczera and Parent, 1998).

ALLOCATION OF POLLUTION CONTROL RESPONSIBILITIES

Recent research related to allocation of pollution control responsibilities has focused on the least-cost (or cost-effective) solutions; however, the cost effective solutions are not necessarily society-wide appealing in policy making due to the concerns of environmental justice and equity. Khadam and Kaluarachchi [2006b] developed an integrated hydrologic and economic model to define the least-cost allocation of abatement efforts of phosphorus loading between contributing sources that also respects equity. The output from the model is used to characterize the trade-off between equity and cost-efficiency in the allocation problem.

The integrated model uses the erosion-scaled export coefficients developed by Khadam and Kaluarachchi [2006a] to model the P-loading from each watershed outlet Equation (2). The economic production of each land use is described here on the basis of per unit area outcome using net present values. The economic production in watershed i is estimated as,

$$\eta = \sum_{j=1}^{n} \eta_j = \sum_{j=1}^{n} \left(w_j A_j - C_j \right) \qquad \dots (6)$$

where w_j is the economic production of land use j (\$/ha) which is calculated from the gross annual revenue of the specific activity divided by the area of the land use; and C_j is the cost of production of land use j (\$) which is assumed constant. Economic production information is typically available from state, country, or national economic production statistics. Cost of production (C_j) is assumed constant to simplify the analysis of the economic model because a constant cost of production will not impact the marginal cost function.

The cost of pollution control is defined as a function of the level of pollution control (M), i.e., the level of reduction in pollution generation. The cost function χ for each watershed is described by,

$$\chi = \sum_{j=1}^{n} \chi_j = \sum_{j=1}^{n} \left(\alpha_j M_j^{\beta_j} A_j \right) \qquad \dots (7)$$

where α_j is a coefficient (\$/ha); β_j is a coefficient (unit less); and n is the number of land use classes in watershed. Equation (7) relates the management cost to the percent reduction in pollution run-off. This cost function is similar to that used by Johansson and Randall [2003]. The major differences between the two cost functions are the use of percent reduction instead of the absolute amount of reduced pollutant and the use of a power relationship instead of a quadratic function. The use of a power relationship allows an additional degree of freedom in the equation such that the curvature of the function is not always quadratic. It should be noted that the underlying assumption of this approach is that infrastructure fixes that would occur at a certain point in time will result in permanent reductions in P-loading. On the other hand, P-control measures that include changes in practices and activities will continue into the future; therefore, their reductions in P-loading will also be permanent.

EQUITY

Khadam and Kaluarachchi [2006b] defined economic equity as the equitable share of the responsibility of environmental protection and restoration and pollution control. Equity is assessed assuming land use as the aggregating unit, i.e., equity among similar land uses. Equity is also assessed using watershed as the aggregating unit, i.e., equity across watersheds. These two characterizations of equity are used to study the impact of the aggregating unit on the outcome of equity. Economic equity is assessed for each watershed and land use class using three criteria; i.e., (1) distribution of pollution control cost, (2) distribution of the ratio of pollution control cost to the gross economic production, and (3) distribution of percentage reduction in P-loadings.

The equity will be measured by the degree of variation among watersheds/land use based on each criterion. The equity measures are formulated such that the highest score for each measure is unity while the lowest score is minus infinity. However, the significant range for each score is between 0 and 1. A score of 1 indicates a uniform distribution, i.e., perfect equity, while a score close to 0.5 indicates a random distribution. Values less than 0.5 indicate a skewed distribution. Equity measure, for this work is defined as distribution of percent economic losses across land uses, measured by the variation in percentage of pollution control costs,

$$EL = 1 - \frac{1}{n} \sum_{j=1}^{n} \left(\frac{\left| \frac{\chi_j}{\eta_j} - \frac{1}{n} \sum_{j=1}^{n} \frac{\chi_j}{\eta_j} \right|}{\frac{1}{n} \sum_{j=1}^{n} \frac{\chi_j}{\eta_j}} \right) \qquad \dots (8)$$

EQUITY-COST EFFICIENCY CURVE

The equity-unconstrained least-cost solution produces the minimum possible cost of control pollution. Therefore, an equity-unconstrained solution receives a cost-efficiency of 100 percent. As the equity requirement increases, the cost also increases resulting in a lower cost-efficiency. Cost efficiency (ε) is measured by the relative incremental cost above that of the equity-unconstrained least-cost solution,

$$\varepsilon = 1 - \frac{\chi_T - \chi_T^{\text{Opt}}}{\chi_T} \qquad \dots (9)$$

where χ_T^{Opt} is the total pollution control cost obtained from the least-cost solution without equity consideration, and χ_T is the total pollution cost obtained from the least-cost solution at a specific level of equity.

For a certain allocation problem, the efficiency-equity curve is developed by estimating cost-efficiencies of the solutions to the problem obtained at several levels of equity. The efficiency-equity curve is then produced by plotting the cost-efficiency values against the corresponding equity scores. The development of the efficiency-equity trade-off curve provides the decision-maker with valuable insight into the cost of equity when considering the allocation of pollution control responsibilities.

CASE STUDY

The study area is the Fishtrap Creek catchment in the lower Nooksack River Watershed located in the northwest corner of Washington State. The Fishtrap Creek catchment is an agriculture catchment of about 95 km^2 (Figure 1). The entire Nooksack River basin area suffers from increased nutrient loadings [Utah State University, 2001] due to heavy agricultural activities consisting of berry cultivation and chicken farms. Phosphorus loading is of concern here because P contributes to water quality problems in the Nooksack River Basin and Puget Sound Bay.

Water quality data required for the analysis include P-application rates for each land use in each watershed, erosion-scaled export coefficients for each land use of each watershed and sediment discharge information for each watershed. P-application rates and an erosion-scaled export coefficient model for the Fishtrap Creek catchment was developed by Khadam and Kaluarachchi [2006a]. Streamflow model of Fishtrap Creek catchment was developed by Khadam and Kaluarachchi [2004]. The economic data needed are economic production data and pollution control costs. Economic production data were obtained from the annual agriculture and animal production statistics of Washington State [Washington Agricultural Statistics Service, 2003]. The cost function for pollution control for each land use class is derived by calibrating Equation (7) to determine α and β values [Khadam and Kaluarachchi, 2006b].

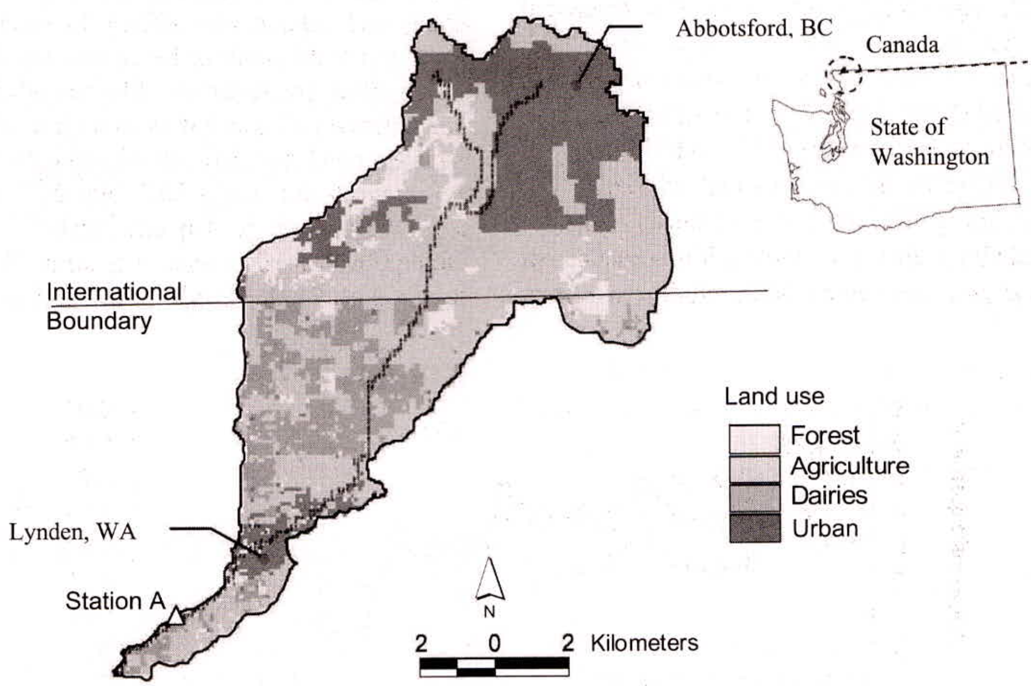

Fig. 1: Layout of the Fishtrap Creek Catchment in Washington State showing the land use distribution

RESULTS AND DISCUSSION

In the export coefficient approach, the annual P application rate for each land use class has to be defined. In addition, the erosion-scaled export coefficient approach developed in this study requires the modeling of sediment discharge from the catchment. The subsequent sections will discuss the estimation of P inputs from each land use and sediment discharge, as well as the calibration procedure to estimate the export coefficients from the observed data.

Sediment discharge was estimate using the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978; Renard *et al.*, 1997). Figure 2 presents the distribution of the P-sediment ratio for samples collected during the period of 1996–1998. For the Fishtrap Creek Catchment, an annual average P-sediment ratio of 2,500 mg/Kg is observed during 1996–1998 (Figure 3). Annual total P is estimated from bi-monthly observations of total P using a period-weighted approach that accounts for magnitude of run-off. A rainfall-run-off model for Fishtrap Creek catchment was described in Khadam and Kaluarachchi (2004).

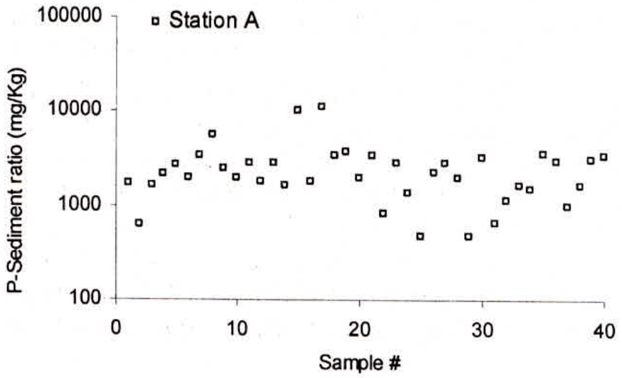

Fig. 2: Observed P-sediment ratio in samples collected from the Nooksack River Basin at the Fishtrap Creek Catchment (Station A)

Calibration of Export Coefficients

Major sources of P to the Fishtrap Creek Catchment are agricultural fertilizers, animal manure, atmospheric deposition, and urban inputs. The P-loading model of the Fishtrap Creek Catchment has four export coefficients, one for each land use class of Table 1. The export coefficients were calibrated using the observed total P loads for the period of 1996–1998. Calibration of export coefficients was performed following the protocol described by Johnes [1996]. For Equation (1), the calibrated export coefficients associated with agriculture, urban, forest and dairies are 0.025, 0.02, 0.02 and 0.035, respectively. The calibrated erosion-scaled export coefficients of Equation (2) for agriculture, urban, forest and dairies are 5.64×10^{-9}, 7.73×10^{-8}, 2.05×10^{-8} and 4.57×10^{-9} (Kg^{-1}), respectively.

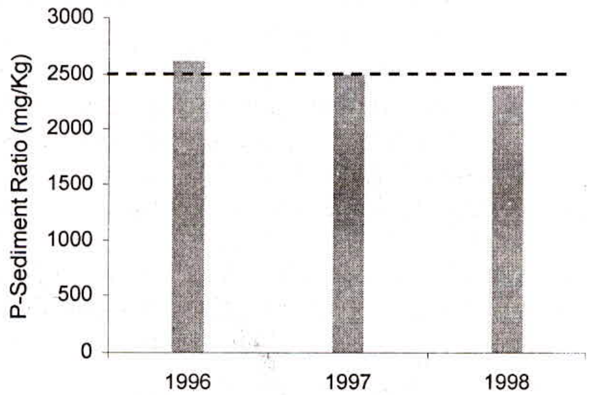

Fig. 3: Annual average P-sediment ratio for the period of 1996–1998 at the Fishtrap Creek Catchment outlet

Table 1: P Application Rates for Each Land Use Class in the Fishtrap Creek Catchment

Land use	*Area (ha)*	*P-application Rate (Kg/ha)*	*Total P-Application (Kg)*
Agriculture	4,896	30.0	151,739
Urban	2,397	1.8	4,267
Forest	836	1.6	1,355
Dairy	1,381	167.8	231,668
Total	9,510		389,029

Figure 4 compares the observed total P for the years 1996 to 1998 with the predictions from the export coefficients (E-predictions) and the predictions from the erosion-scaled export coefficients (*K*-predictions). It shows significant improvements in the prediction of annual P loadings when hydrologic variability is considered. The observed P loadings during the 1996–1998 period show considerable variability, although the land use and P application rates remained unchanged during this period. Despite the apparent variability in the total annual P loadings, the P-sediment ratio remained almost constant for the same period (see Figure 3). This observation suggests that the P-sediment ratio is linked primarily to the land use pattern and P management practices, i.e., effective soil-P concentration. Since these two factors remained unchanged, the P-sediment ratio remained unchanged too. Hence, we can expect that the changes in land use and management practices would be reflected in the

P-sediment ratio; however, the response of the P-sediment ratio to changes in land use and management practices may be lagged because of the role of P-soil reservoirs.

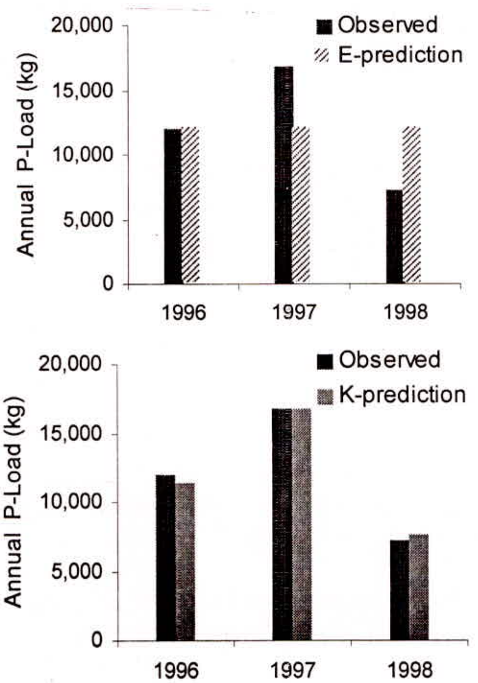

Fig. 4: Observed annual P load from the Fishtrap Creek Catchment for the period of 1996 to 1998 compared to the predicted annual P loading from the export coefficients (*E*-prediction), and from erosion-scaled export coefficients (K-prediction)

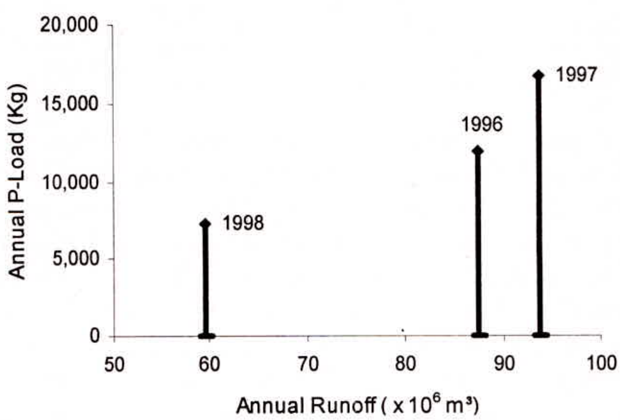

Fig. 5: Relationship between the annual P loading and the annual run-off observed in the Fishtrap Creek from 1996 to 1998

The variability in the observed P loadings can be best explained by the variability in run-off shown in Figure 5. These results show that the volume of run-off affects the amount of sediment discharge from a catchment, which in turn dictates the total P loadings. Hence, hydrologic variability can explain the variability in P loadings despite the fact that land use and management practices did not change. Since the erosion-scaled export coefficient model can capture the effect of hydrologic variability through the sediment discharge, its predictions of the P loadings managed to capture the variability in the observed P loadings as seen in Figure 4. Erosion-scaled export coefficients, thus, prove to be superior to the traditional export coefficients of Equation (1) in explaining the variability in observed P loadings, while maintaining the simplicity and low data requirements that are characteristic of the export coefficient approach.

Uncertainty of Export Coefficients

To assess the uncertainty of the export coefficient model developed for the Fishtrap Creek Catchment, the Bayesian-based procedure described earlier in the methodology section was used. The prior probability distribution of export coefficients (E) was assumed to be uniformly distributed between 0 and 0.2, i.e., no more than 20 percent of the P applied will be exported from any land use (USEPA, 1980). Figure 6 presents the predicted posterior probability density functions of the export coefficients (E) for each land use. Posterior probability density functions for the erosion-scaled export coefficients (K) were generated using a prior uniform distribution in the range [0, 10^{-7}] and shown in Figure 7.

For urban and forest land uses, Figure 6 shows that their posterior distributions are essentially the same as the prior distributions. These observations indicate that the observed data for the period 1996–1998 do not provide new information that would help identify export coefficients for these two land use classes. As a result, any value between 0 and 0.2 is an equally feasible value for forest and urban export coefficients.

For erosion-scaled export coefficients (K), updated parameter uncertainty is shown in Figure 7. Comparing Figure 7 to Figure 6, it is evident that the posterior probability distribution for both export coefficients (E) and erosion-scaled export coefficients (K) are similar. Equation (3) describes K as a linear function of E. Hence, the posterior probability distributions of E and K should be similar and are proven here with this results. An important consequence of this trivial fact is that hydrologic variability and parameter uncertainty are two independent features of overall model uncertainty. Although the erosion-scaled export coefficient approach improved the prediction accuracy of the annual P load (see Figure 4) by incorporating hydrologic variability, its parameter uncertainty remained unchanged.

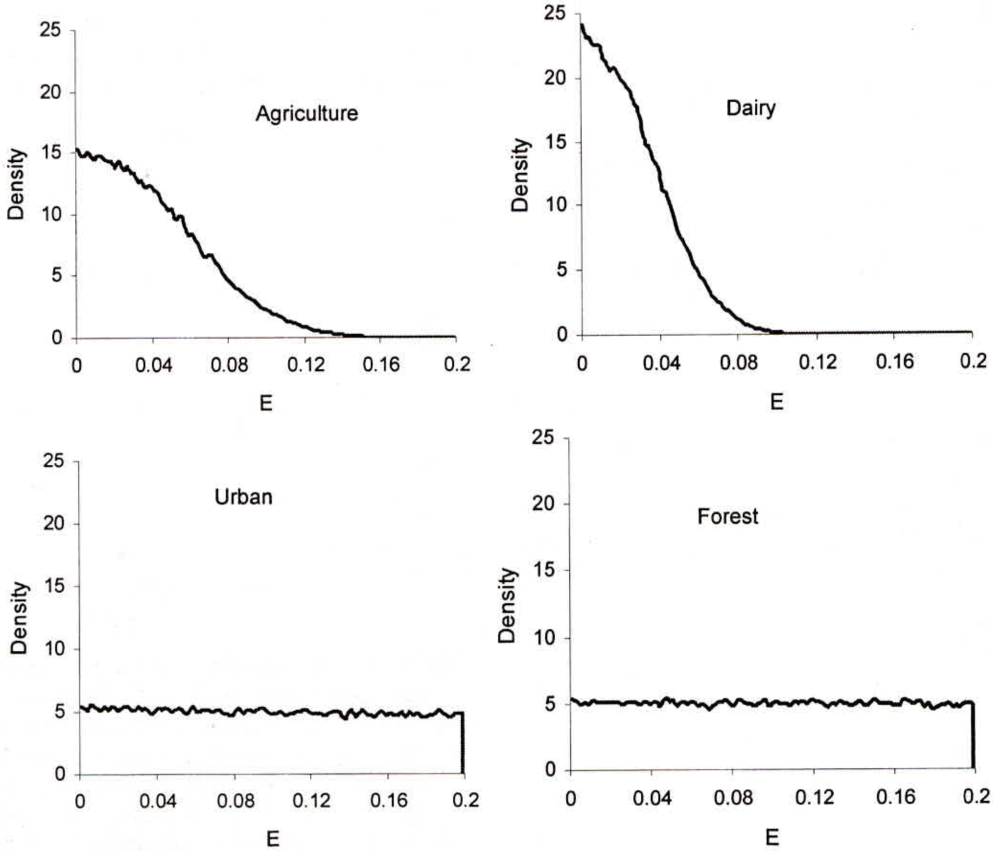

Fig. 6: Predicted posterior density functions of export coefficients using a uniform prior distribution in the range [0.0, 0.2]

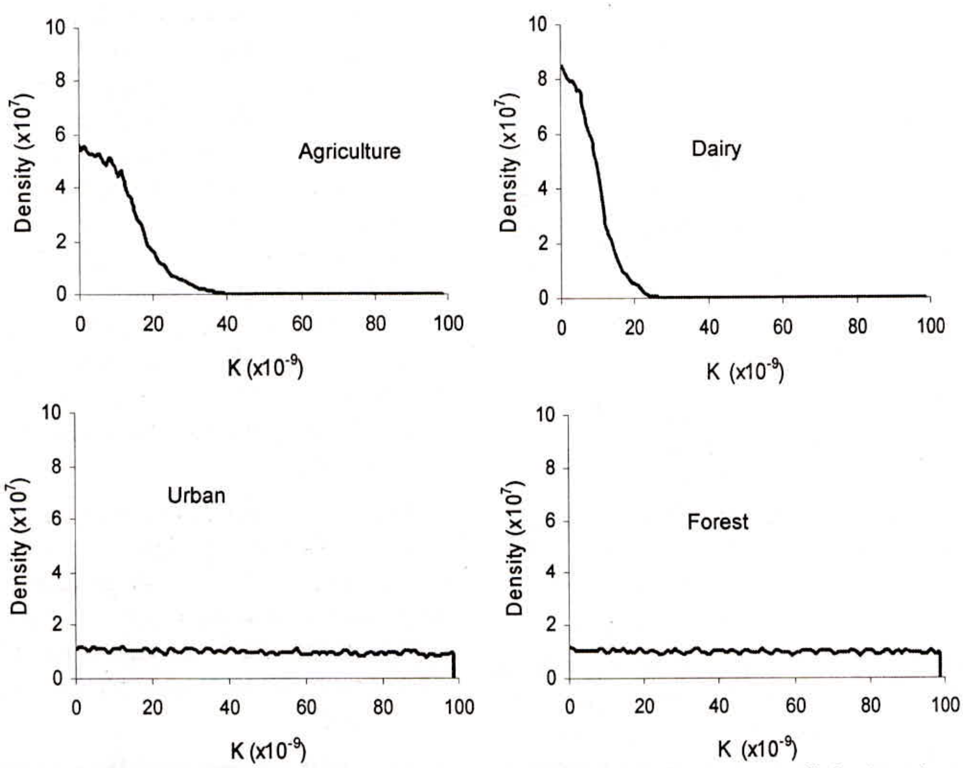

Fig. 7: Predicted posterior density functions of erosion-scaled export coefficients using a uniform prior distribution in the range [0.0, 10^{-7}]

Cost of Pollution Management

Figure 8 presents the cost of pollution control in the Fishtrap Creek catchment for different levels of pollution control at 50 percent reliability without equity considerations. The 50 percent reliability is established by defining $\rho = 0.5$ in the probabilistic water quality constraint (Equation 2). The shape of the curve in Figure 8 reflects the cost-effectiveness of pollution control efforts. The initial part of the curve has a steep slope (SL 1), compared to the flatter slope at the end of the curve (SL 2), i.e., high cost-effectiveness for SL 1 and low cost-effectiveness for SL 2. This shape of the cost curve is decided by the nature of the cost function of pollution control at each pollution source. In this study, we employed a power cost function (see Equation 7) which follows the shape of the curve in Figure 8.

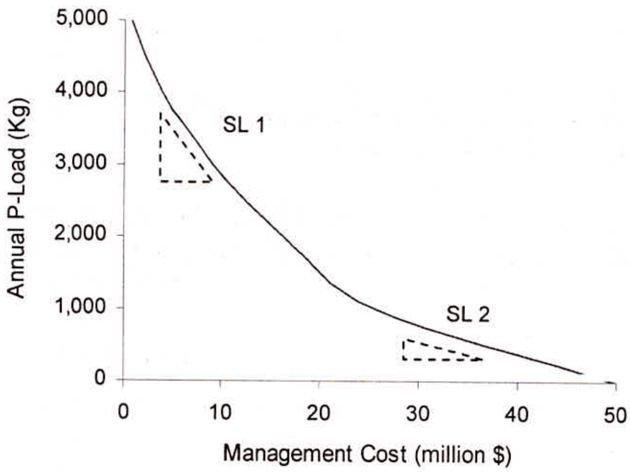

Fig. 8: Cost of managing annual P-loading in the Fishtrap Creek catchment to different annual P-levels. Costs were obtained from the least-cost equity-unconstrained solution at solution at 50 percent exceedance probability

Efficiency-Equity Trade-Off Curve

The efficiency-equity trade-off curve for reducing P run-off to an annual amount of 3,000 Kg at 50 percent reliability is shown in Figure 9. Development of the efficiency-equity curve requires the estimation of cost efficiencies of the solutions at several levels of equity based on the equity measure EL, i.e., the distribution of percent economic losses between land uses.

The cost-efficiency axis of Figure 9 represents the deviation from the equity-unconstrained solution from each level of equity. For example, a cost efficiency of 80 percent indicates that the allocation of pollution responsibilities will cost 20 percent more compared to the equity-unconstrained least-cost allocation which is the 100 percent efficiency solution. Similarly, an equity score of one means that the ratio of pollution control costs to gross economic production is equal among all land use classes.

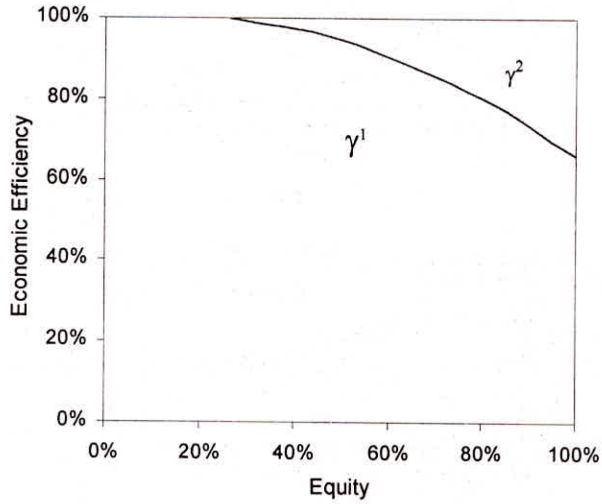

Fig. 9: Equity cost efficiency curve describing the trade-off between equity and cost efficiency for reducing the expected annual P-loading in the Fishtrap Creek catchment to 3,000 Kg. The least-cost equity-unconstrained solution to achieve this level of P-management is $10 M (refer to Figure 8), which corresponds to 100 percent cost efficiency. Equity score is defined by Equation 8

Figure 9 shows two regions, γ^1 and γ^2. The region γ^1 is the space of feasible solutions while region γ^2 is the space of non-feasible solutions. As a result, it is not feasible to achieve both maximum cost efficiency and equity score at the same time in the Fishtrap Creek catchment when attempting to maintain P-run-off of 3,000 Kg because that point falls in the non-feasible space or γ^2. It should be noted that any point on the efficiency-equity curve describes the least-cost solution for a given equity level or the maximum equity that can be achieved under a given cost efficiency. Therefore, there may be multiple solutions or choices of cost-efficiency values for a given equity or similar multiple choices of equity values for a given cost efficiency.

This observation has an important implication in the political process of allocation of pollution control responsibilities. Since this process typically involves negotiations between different stakeholder groups, it is important that the final negotiated solution falls on, or close to, the efficiency-equity curve, such that optimality is achieved.

Figure 10 presents the effect of increasing environmental stringency, i.e., the impact of reducing the allowable annual P-load from 3,000 Kg to 1,000 Kg.

Table 2: Effect of Increasing Stringency of Water Quality Constraint on the Trade-off between Equity and Cost Efficiency (refer to Figure 10)

	Solution Y		*Solution Y**	
Expected P-load (L_T)	3,000 Kg		1,000 Kg	
Cost efficiency (ε)	90%		90%	
Equity (q)	63%		95%	
Total pollution control cost (χ_T)	$9.65 M		$33.0 M	
	Dairy	*Agriculture*	*Dairy*	*Agriculture*
P-load reduction	46%	51%	80%	86%
Pollution control cost (χ)	$5.48 M	$4.17 M	$13.20 M	$19.80 M
Economic production (η)	$18.74 M	$31.05 M	$18.74 M	$31.05 M
Economic loss (χ/η)	29%	13%	70%	64%

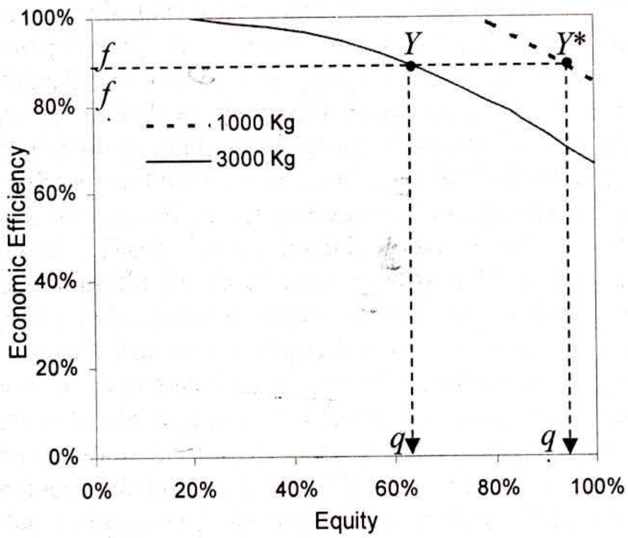

Fig. 10: Effect of increasing stringency of water quality constraint on the trade-off between equity and cost efficiency for P-management in the Fishtrap Creek catchment. Equity score is defined by Equation 8

Consider the two points, *Y* and *Y**, located on the 3,000 and 1,000 Kg curves, respectively. Although the cost-efficiency values for solutions *Y* and *Y** are similar ($\varepsilon = \varepsilon^*$), it should be noted that the actual pollution control costs for each solution is different. In fact, the management cost for solution *Y** is more expensive than that of *Y* due to the additional reduction in P-load required (see Table 2).

Although both solutions (*Y* and *Y**) result in similar cost efficiencies, these solutions represent different levels of equity. Figure 10 shows that for the same level of cost efficiency, equity increases with the increased reduction of pollution load, i.e., increased stringency of water quality constraints. The solution *Y* calls for a pollution reduction of 46 and 51 percent from dairies and agriculture, respectively. The solution *Y** calls for a pollution reduction of 80 and 86 percent from dairies and agriculture, respectively. Increasing stringency of water quality constraints requires more pollution reduction from land uses, which in turn increases cost of pollution control. The increased control costs, coupled with a fixed economic production, result in smoothing of the variations in the distribution of the ratio of pollution control cost to the gross economic production across land uses. Tighter levels of control generally make cost efficiencies easier to obtain at any given level of equity due to convex nature of cost (see Figure 10).

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