WORKING PAPER
2017-06

REPA

Resource Economics
& Policy Analysis
Research Group

Department of Economics
University of Victoria

Maximizing Returns from Payments for Ecosystem Services: Incorporating Externality Effects of Land Management

Mark E. Eiswerth and G. Cornelis van Kooten

November 2017

For copies of this or other REPA working papers contact:

REPA Research Group
Department of Economics
University of Victoria PO Box 1700 STN CSC Victoria, BC V8W 2Y2 CANADA
repa@uvic.ca
Fax: 250.721.6214
http://web.uvic.ca/~repa/

Maximizing Returns from Payments for Ecosystem Services: Incorporating Externality Effects of Land Management

Mark E. Eiswerth
Department of Economics
University of Northern Colorado

G. Cornelis van Kooten
Department of Economics
University of Victoria

October 25, 2017

Abstract

Given the expansion of payments for ecosystem services (PES) programs worldwide, two relevant issues are: (1) determination of efficient allocations of payments among land managers, and (2) how this might change in the presence of cross-manager externalities, whereby when one manager is paid to implement a best management practice (BMP) to enhance an ecosystem service, there is an impact (either positive or negative) on the BMP cost effectiveness of other land managers. For a given context such externalities could be negative on the whole if diminishing returns dominate, or positive if mechanisms such as 'social diffusion' dominate. This manuscript analyzes how a PES planner should optimally allocate payments among land managers, depending on whether expected cross-manager externalities are negligible, negative, or positive. We employ 1) static analysis to shed initial intuitive light, 2) optimal control methods to gain insights on the dynamics of the problem, and 3) stochastic dynamic programming to determine optimal PES funding strategies with specific application to water-based ecosystem services in a watershed. This contributes to the literature by identifying dynamically optimal PES payment patterns, whether externalities exist or not. In addition, the results show how the allocation of PES payments should change when one accounts for externalities induced by the program. Because such spillover impacts have not been addressed previously in a rigorous way, this treatment provides useful value added for PES design and implementation.

Key words: payments for ecosystem services; water quality; best management practices; externalities; dynamic optimization; uncertainty; principal-agent problems.

JEL Categories: Q25, Q57, C61

INTRODUCTION

Ecosystem services have been defined in different ways, but most definitions focus on outcomes of ecosystem processes that directly influence human health and welfare, or that maintain the quantity and quality of various ecosystem goods. Daily (1997) offers the following definition: “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life. They maintain biodiversity and the production of *ecosystem goods,* such as seafood, forage, timber, biomass fuels, natural fiber, and many pharmaceuticals, industrial products, and their precursors” (see also Brown et al. 2007). This makes clear a distinction between ecosystem services, which consist of “actual life-support functions, such as cleansing, recycling, and renewal, … [that] confer many intangible aesthetic and cultural benefits as well” (Daily 1997), and ecosystem goods that are used in production or final consumption.

Other definitions of ecosystem services differ in various ways, but can be quite concise; e.g., “the aspects of ecosystems utilized (actively or passively) to produce human wellbeing” (Fisher et al. 2008) and “the benefits people obtain from ecosystems” (Millenium Ecosystem Assessment 2003). While popular definitions are used to educate the public about ecosystems, they are “excessively broad, easy to misinterpret, and are of limited use in more detailed analysis of [ecosystem service] payment schemes” (Farley and Costanza 2010).

Increasing attention has been focused in recent years on how payments for ecosystem services (PES) can be conceptualized, designed, implemented and evaluated (Daniels et al., 2010; Engel et al. 2008; Wunschler et al. 2008; Farley and Costanza 2010; Kemkes et al. 2010; Norgaard 2010; Kolinjivadi et al. 2014; Sommerville et al., 2009). In this regard, Goldman-Benner et al. (2012) define PES as programs that “include voluntary transactions where well-defined environmental (or ecosystem) services (or land uses likely to secure those services) are bought by a minimum of one service buyer, from a minimum of one service provider, if and only if the service provider continuously secures service provision (conditionality).” Other practices sometimes considered as desirable include *additionality* (the buyer of ecosystem services only pays the seller for actions/practices that would not otherwise take place) and the *exclusion or limitation of side objectives* (the program excludes or minimizes the pursuit of additional objectives) (Engel et al. 2008; Goldman-Benner et al. 2012). Nonetheless, debate continues about the desirability of strictly adhering to such conditions (Goldman-Benner et al. 2012).[1]

The application in the current study is to the use of payments for water-based ecosystem services, for which we employ the acronym PWES. We recommend this ‘term’ in part because, during a period of rapid growth in PWES programs, different authors have used a variety of other terms to refer to such programs. For example, Forest Trend’s Ecosystem Marketplace refers to the programs as an “investment in watershed services” (Bennett and Carroll 2014). Lin et al. (2013) employ the term “payments for improving ecosystem services at the watershed scale (PIES-W),” and Lin (2012) considers PIES-W to be an “integrated ecosystem management approach” that gives watershed stakeholders incentives to assist in management actions, by facilitating “demand-side payments from downstream payers to finance supply-side activities [expected to enhance ecosystem services] conducted by upstream payees.” There are several different types of programs that one may consider to

[1] This debate is found in discussions of the role that avoided deforestation and degradation, and sustainable forest management and conservation, play in mitigating climate change (e.g., see Malmsheimer et al. 2011; Buttoud 2012).

constitute PWES, such as bilateral agreements, water funds, instream buyback approaches, trading and offset schemes. We do not distinguish among overarching terms, such as PWES and PIES-W, since most such definitions tend to be broadly similar. Rather, we employ the term PWES and, where our analysis relates more generally to ecosystems rather than solely watershed-based ecosystems, we use PES.

Our primary focus is on two questions relevant to the design and implementation of PES programs. First, what is the optimal allocation of payments among different land managers, each of whom is possibly willing to participate in a program by acting as a seller of ecosystem services? Second, how does the optimal allocation change when there are various types of cross-producer, or cross-land manager (hereafter just cross-manager), external effects in the provision of ecosystem services? By the term 'cross-manager external effects' we mean the following: when one land manager implements a best management practice (BMP) designed to produce an ecosystem service (ES), this BMP affects the 'marginal ecosystem service product' (MESP) of other land managers in the watershed. We define MESP as the additional amount of an ES that is provided by one more unit of effort (or expenditure) made by a given land manager on the BMP(s) designed to produce the ES. A cross-manager external effect is thus the externality imposed by one ES producer (land manager) on another.

Why would one expect cross-manager external effects to exist? There are a variety of possibilities. For example, standard economic theory would suggest that any given land manager's MESP might decline as other land managers provide more of the same ecosystem service. For instance, suppose that the BMP in question involves implementing buffer strips along a river or stream, the edge-of-property metric is phosphorus loadings, and the ES under consideration is the provision of water quality and habitat conditions suitable for aquatic species. Then it may be the case that the additional ES yielded by buffering the last 10% of the total streamside linear feet is less than that generated by buffering the first 10%; if the MESP declines in this way, then there is a negative cross-manager external effect at work. As one land manager deploys more of a BMP, the effectiveness of one more unit of another land manager's BMP implementation may decline, *ceteris paribus*.

In contrast, there also may be positive cross-manager external effects at work. For example, a PWES program may have positive benefits via social spillover impacts. That is, when one land manager adopts a particular BMP as the result of a PWES payment, other land managers in the watershed also implement the practice without a subsidy (Goldman-Benner et al. 2012). There are several possible transmission mechanisms why one land manager's BMP application may increase the probability that a neighboring land manager implements the same BMP. First, when one land manager implements a BMP, it may improve other land managers' levels of knowledge regarding that BMP – diffusion by imitation. Second, by observing and perhaps discussing the execution of a BMP with other land managers, the effectiveness of copying it is enhanced. Put simply, initial adoption of the BMP in the watershed allows other land managers to learn about the BMP and how best to implement it (improvements in knowledge and skill comprise the positive spillover). As knowledge and skills related to the BMP increase, it may become less expensive to implement the BMP – learning by doing that reduces costs. Finally, as more land managers throughout the watershed adopt the BMP, scale-related reductions in input prices could occur; for example, the establishment of a local distribution center for BMP inputs may reduce transportation costs.

For any given watershed, it is reasonable to expect that, in general, there would be a number of both positive and negative cross-manager external effects at work as described above. However, it is also reasonable to expect differences across watersheds in terms of the nature and extent of these individual effects. Summing the various positive and negative externalities yields what we refer to as the net cross-manager external effects. For any given BMP and ecosystem service, some watersheds may exhibit positive net cross-land manager external effects while others are characterized by net negative effects.

Intuitively, one would predict that the characteristics of a watershed, in terms of net cross-manager external effects, should have implications for the optimal targeting of PWES payments. For example, if conservation practices implemented by some land managers are expected to yield substantial positive externalities for others, it may be optimal to target payments toward such managers. Unfortunately, current practice related to PWES often fails to consider externalities in their entirety. Therefore, there is a need for research to elucidate the importance of and ways of accounting for these factors in program design and operation.

The remainder of this paper is organized as follows. First, we employ a static model to illustrate the nature of the issues discussed above. Next, we use optimal control theory to illuminate the nature of the issues when viewed in a dynamic context. This is followed by a numerical application using stochastic dynamic programming that illustrates how these issues might be investigated for a specific watershed. Finally, we conclude by discussing the implications of our analysis for policy as well as directions for further research.

MODELS

A Simple Static Framework

To gain initial insights, we begin with static analysis that considers one watershed, two land managers, and one ES, although the analysis can be expanded to include multiple environmental services (e.g., using a weighted index of multiple ES), watersheds and land managers. In addition, as discussed more fully in the context of our dynamic model, it is useful to think of the two land managers as representing two categories of managers, that is, as applying to a case where there are multiple land managers that a PES planner has sorted into two categories, *i* and *j*, based on a portfolio of relevant characteristics. We define a given land manager's baseline management practices as the current practices prior to any adjustments made in response to payments received through a PWES program.

Upon receipt of payments, a land manager implements BMPs that increase the level of the ES in the watershed through a series of steps viewed simply as follows:

- *Step 1: ΔBMP Effort (Land Manager Inputs) → Δ Edge-of-Property (EOP) Metric*
- *Step 2: ΔEOP Metric → Δ Watershed-wide Metric*
- *Step 3: Δ Watershed-wide Metric → Δ Level of Ecosystem Service (ES)*

ΔBMP Effort represents changes in expenditures/effort devoted to BMPs (e.g., changes in tillage practices, application rates of fertilizers or pesticides, buffer strips, etc.). An example of a *ΔEOP metric* would be total annual loadings of phosphorus from property *i*, where a corresponding *Δwatershed-wide metric* would be average phosphorus concentrations and *ΔES* would be change in habitat quality. While each of the forgoing steps are written as general and partial (i.e., assuming *ceteris paribus*), we focus on the likely chain of events as a result of PWES, initially holding other factors constant. Additional layers of complexity that include multiple causal factors, dynamics and uncertainty may be added subsequently.

Case 1: No net cross-manager external effects in the provision of ES.

First consider the case where net cross-manager (agent) external effects are zero or, alternatively, where external effects are not taken into consideration by the PES planner (principal). Assume that the principal is interested in the level of the ES in the watershed that will exist following payments to support the adoption of ES-enhancing BMPs. In the simplest case, since BMP implementation will require some amount of time to affect the level of the ES, one may think of the principal (planner) as considering the impact that a BMP implemented today will have on tomorrow's level of the ES:

$$S_i = b_i + f_i(E_i), \tag{1}$$

where S_i is the level of the ecosystem service that is provided in the next time period by land managed by i (denoted M_i), b_i is the baseline level of the ES provided by M_i if the BMP is not implemented, and $f_i(E_i)$ is the additional level of the ES provided by M_i if she spends E_i on BMPs, where $f_i'(E_i) > 0$, $f_i''(E_i) < 0$. In this application, E_i denotes expenditure devoted to implement one or more BMPs designed to provide a higher level of the ES.

For the simplest case of two land manager categories i and j, the PES planner's static constrained maximization problem may be viewed as:

$$\text{Max } [V(S_{\mathrm{i}} + S_{\mathrm{j}})] \qquad \text{s.t.} \qquad E_{\mathrm{i}} + E_{\mathrm{j}} = E \tag{2}$$

where V is the per-unit value of the ES provided in the watershed, and E is the total expenditures that the planner chooses to implement ES-producing BMPs in the watershed. For example, the managing board of a water fund may decide that it wishes to provide \$$E$ to implement one or more BMPs, and how to allocate that funding across land managers.

The first-order conditions to the constrained maximization problem (2) is

$$f_i'(E_i) = f_j'(E_j),\ \forall\, i,j, \tag{3}$$

which expresses the standard condition that the marginal benefits of investment in BMPs should be equalized across land managers. We can refer to $f_i'(E_i)$ as the marginal ecosystem service product of *M_i's* effort/expenditure on BMPs ($MESP_i$). Since V is assumed to be constant, (3) implies that not only marginal products but also marginal benefits are equalized across land managers.

As a simple illustration, consider the following specification for f_i that satisfies the standard assumption of diminishing marginal returns to expenditure:

$$f_i(E_i) = b_i\, E^{-\frac{1}{2}}\, E_i^{\frac{1}{2}}. \tag{4}$$

Given (4) and the constrained maximization problem (2), the first-order conditions would be:

$$b_i\, E_i^{-\frac{1}{2}} = b_j\, E_j^{-\frac{1}{2}}. \tag{5}$$

If the intercepts were the same for both land managers' ES provision functions, expenditures would be equated across parcels (because the exponential terms are the same). In general, one would not expect this to be the case, so we subsequently relax this assumption.

Case 2: Net negative cross-manager external effects.

In this case all assumptions remain the same as in Case 1, except that net cross-manager external effects are assumed to be negative. While there are expected to be both positive and negative cross-manager external effects at work, we assume the negative external effects outweigh the positive ones. This results in the following new terms pertinent to the ES provision function, which for this case we denote as $g_i(E_i)$:

$$g_i'(E_i) = f_i'(E_i) + \delta_{ij}, \tag{6}$$

where $\delta_{ij} < 0$ represents the change in the marginal product (slope) of *M_i's* ES provision function that occurs due to the other land manager's (M_j*'s*) implementation of BMP. Thus, in Case 2, the ES provision function for M_i becomes:

$$S_i = b_i + f_i(E_i) + \delta_{ij} E_i \tag{7}$$

Given (7), the generalized first-order condition (FOC) to the constrained maximization problem (2) becomes:

$$f_i'(E_i) = f_j'(E_j) + (\delta_{ji} - \delta_{ij}),\ \forall\, i,j, \tag{8}$$

where $(\delta_{ji} - \delta_{ij})$ may be positive, negative or zero depending on the relationships between the two cross-manager external effect terms, which are both negative in Case 2. For example, suppose that $|\ \delta_{ij}\ | > |\ \delta_{ji}\ |$, that is, the reduction in $MESP_i$ imposed by M_j*'s* BMP ($|\delta_{ij}|$) exceeds the reduction in $MESP_j$ imposed by M_i*'s* BMP ($|\delta_{ji}|$). Then, $(\delta_{ji} - \delta_{ij}) > 0$ and so $f_i'(E_i)$ should optimally be set greater than $f_j'(E_j)$, which means that $f_i(E_i)$ should be set lower than if there were no net cross-manager external effects (i.e., $\delta_{ji} - \delta_{ij} = 0$) as in Case 1. In other words, the land manager (*i* in this case) whose *MESP* becomes relatively more depressed as a direct result of the other land manager's provision of ES (i.e., their *MESP* is more adversely sensitive to the other land manager's actions), optimally should provide less of the ES by implementing a smaller level of BMPs than she otherwise would in the absence of net negative cross-manager external effects. If $|\ \delta_{ij}\ | < |\ \delta_{ji}\ |$, on the other hand, then M_i optimally should implement a higher level of BMPs than otherwise compared to Case 1.

Case 3: Net positive cross-manager external effects.

In Case 3, all assumptions remain the same as in Case 1, except that net cross-manager external effects are positive; Case 3 is the opposite of Case 2. Thus, in (6), $\delta_{ij} > 0$, representing an increase in the slope of M_i*'s* ES provision function as a result of the other land manager's (M_j*'s*) implementation of BMPs. The generalized FOCs (8) remain the same; it is still the case that $(\delta_{ji} - \delta_{ij})$ may be positive, negative or zero depending on the relationships between the two cross-manager external effect terms. The difference from Case 2, however, is that now both δ_{ij} and δ_{ji} are positive. Suppose that $\delta_{ji} > \delta_{ij}$ so the increase in $MESP_j$ imposed by M_i*'s* BMP (δ_{ji}) exceeds the increase in $MESP_i$ imposed by M_j*'s* BMP (δ_{ij}). Then, $(\delta_{ji} - \delta_{ij}) > 0$ and $f_i'(E_i)$ should optimally be set greater than $f_j'(E_j)$, which means that $f_j(E_j)$ should be set higher than if there were no net cross-manager external effects (i.e., $\delta_{ji} - \delta_{ij} = 0$) as in Case 1 above. In other words, the land manager (*j* in this case) whose *MESP* increases relatively more as a direct result of other land managers' provision of the ES via BMPs (i.e., their *MESP* is more favorably sensitive to another land manager's actions), optimally should provide more of the ES than she would in the absence of net positive cross-manager external effects. If $\delta_{ji} < \delta_{ij}$, on the other hand, then M_i optimally should implement a higher level of BMPs than otherwise (i.e., as compared to Case 1).

A Dynamic Control Framework

Now consider the implications of incorporating cross-manager external effects explicitly in a dynamic framework. We begin by developing a simple, one state variable and one control variable, dynamic optimization model. An ecosystem service can be modeled as either a stock or flow, but we treat it as a stock that then yields a flow of environmental benefits to society. For example, aquatic habitat of a certain quality may be thought of as a stock that then yields a stream of benefits (both active and passive uses) connected with the existence of fish and other animal species.

Given this perspective, and again assuming two land manager types as previously, one may model the stock of the ecosystem service at any given time, S_t, as changing instantaneously according to the following equation of motion:

$$\dot{S} = f_1[E_1(t)] + f_2[E_2(t)] - \alpha S(t), \tag{9}$$

where $f_i[E_i(t)]$ is the addition to the stock of the ecosystem service (S) provided by land manager i's expenditure E_i on BMP in period t; $f_i'(E_i) > 0$, $f_i''(E_i) < 0$ as in the static framework; and $\alpha > 0$ is the mean rate of natural decay or depreciation of the watershed's ecosystem service. The term 'rate of natural decay' does not imply that the rate is not affected by anthropogenic activities; rather, it is the rate of decay that would occur if the land managers were not to implement BMPs. The mean rate of natural decay of the ecosystem service stock, α, will depend on the physical, chemical, hydrological and biological characteristics of the watershed. In aggregate, the process of natural decay is assumed to be given by:

$$S_{\mathrm{t}} = S_0 \, e^{-\alpha t}, \tag{10}$$

where S_0 is the value of S at time $t = 0$.

Consider the objective of maximizing the future discounted flow of monetized ecosystem services, less the payments made to land managers to fund the implementation of BMPs that augment the stock S. Recall that we assume the principal (planner) has sorted multiple land managers (agents) into two categories based on a portfolio of characteristics, including the expected nature (signs and magnitudes) of their cross-manager external effects. Therefore, in referring to land managers 1 and 2 (M_1 and M_2), we actually are referencing two categories of land managers.

Rather than incorporating E as the total expenditure that the principal is able to spend on incentivizing land managers, E_1 and E_2 denote the *shares* of total BMP effort that are allocated to M_1 and M_2, respectively. As a result, the principal will choose the value of E_1 over time with E_2 the associated residual of that decision.

Now consider cross-manager external effects. In the static model, we assumed that the terms δ_{ij} and δ_{ji} were constants, but we expect the magnitude of the externality produced by a land manager to be a function of the level of BMP implemented by that manager. In particular, as M_j's choice of the level of BMP (E_j) increases, the level of the externality imposed on M_i will increase as well. In general terms, let $h_{ij}(E_j)$ denote the net external effects on M_i*'s* ES provision function as a result of M_j*'s* implementation of BMP. Then, for a situation in which the external effects from M_1's actions on M_2 are net positive as E_1 increases, the positive external impact on M_2 will become a larger positive number, so that $h_{21}(E_1)>0$, $h'_{21}(E_1)>0$. Likewise, for a situation in which the external effects on M_1 by M_2's actions are net positive; then, as E_2 increases, the positive externality to M_1 will increase, i.e., $h'_{12}(E_2)>0$. Since E_2 is treated as a residual of the choice of E_1, $h'_{12}(E_1) < 0$.

The signs for total and marginal effects will be different for the case of net negative cross-manager external effects. As E_1 increases, the negative externality imposed on M_2 will increase (in absolute terms), so that $h_{21}(E_1) < 0$, $h'_{21}(E_1) < 0$. Likewise, for a situation in which the external effects imposed on M_1 by M_2 are net negative, as E_2 increases, the negative externality imposed on M_1 will become larger in absolute value, so that $h'_{12}(E_2) < 0$.

The principal's problem, accounting for external effects, may be formulated as an infinite horizon, continuous-time, constrained optimization problem as follows:

$$\text{Maximize} \int_{o}^{\infty} [V(S_t) - C_1(E_{1t}) - C_2(E_{1t})]\, e^{-rt}\, dt, \tag{11}$$

$$\text{subject to: } \dot{S} = f_1(E_{1t}) + f_2(E_{1t}) + [h_{12}(E_{1t})\times(E_{1t})] + [h_{21}(E_{1t})\times(1-E_{1t})] - \alpha S_t \tag{12}$$

$$S(t_0) = S_0,\ \lambda_1(t_1) = 0 \tag{13}$$

$$1 - E_1 - E_2 \geq 0 \tag{14}$$

The current-value Hamiltonian for this problem is:

$$H = V(S_t) - C_1(E_{1t}) - C_2(E_{1t})$$
$$+ \lambda_1\{ f_1(E_{1t}) + f_2(E_{1t}) + [h_{12}(E_{1t})\times(E_{1t})] + [h_{21}(E_{1t})\times(1-E_{1t})] - \alpha S_t \} \tag{15}$$

and, since the dynamic optimization problem includes static budget (or effort) constraints that must be met in every period, the augmented Lagrangian function is:

$$L = V(S_t) - C_1(E_{1t}) - C_2(E_{1t})$$
$$+ \lambda_1\{ f_1(E_{1t}) + f_2(E_{1t}) + [h_{12}(E_{1t})\times(E_{1t})] + [h_{21}(E_{1t})\times(1-E_{1t})] - \alpha S_t \}$$
$$+ \lambda_2 [1 - E_{1t} - E_{2t}] \tag{16}$$

where r is the (social) rate of discount; λ_1 is the current value co-state variable associated with the current-value Hamiltonian and denotes the shadow price of the ES stock; λ_2 is the constant multiplier associated with the BMP effort 'endowment share' constraint; t_1 is the free terminal point (endogenously determined ending time at which λ_1 becomes zero); $C_1(E_1)$ is M_1's total cost of implementing the BMP, $C_1'(E_1)>0$, $C_1''(E_1)>0$; $C_2(E_1)$ is M_2's total cost of implementing the BMP, $C_2'(E_1)<0$, $C_2''(E_1)>0$; and where E_1 (the control variable), E_2 and S are as previously defined. Note that M_2's costs are expressed as a function of E_1 since E_2 is a residual in this share model.

The necessary conditions for a solution to this problem (suppressing all time subscripts) are:

$$\lambda_1 f_1'(E_1) + [-h_{21}(E_1) + h_{21}'(E_1)\times(1-E_1)] + \lambda_1 f_2'(E_1) + [h_{12}(E_1) + h_{12}'(E_1)\times E_1]$$

$$= C_1'(E_1) + C_2'(E_1) + \lambda_2 \qquad (17)$$

$$V'(S) = \lambda_1(\alpha + r) - \dot{\lambda}_1 \qquad (18)$$

$$\dot{S} = f_1(E_1) + f_2(E_1) + [h_{12}(E_1)\times(E_1)] + [h_{21}(E_1)\times(1-E_1)] - \alpha S \qquad (19)$$

$$S(t_0) = S_0,\ \lambda_1(t_1) = 0 \qquad (20)$$

$$1 - E_1 - E_2 \geq 0;\ \lambda_2 \geq 0;\ \lambda_2(1 - E_1 - E_2) = 0 \qquad (21)$$

Condition (17) indicates that, for an interior solution, the control variable E_1 should be set at a level such that the marginal benefits of BMP expenditures (including the marginal positive and negative benefits embedded in a series of terms representing the cross-manager externalities resulting from BMP investments) should equal the marginal costs of the BMP investments plus the shadow price of available but unspent BMP endowment funds. Condition (18) indicates the optimal path of the ES stock shadow price over time, which depends on the marginal value of the ES stock, the natural rate of decay or depreciation of the ES stock, and the social rate of discount. Conditions (19), (20) and (21) are, respectively, the state equation, the endpoint conditions, and the Karsh-Kuhn-Tucker conditions related to the BMP total effort constraint.

To derive the steady-state condition for E_1, use (17) and (18) to solve for λ_1 and $\dot{\lambda}_1$. First, rearrange (17) to solve for λ_1 as follows:

$$\lambda_1 = [-(l_{21}(E_1) + m_{12}(E_1)) + C_1'(E_1) + C_2'(E_1) + \lambda_2] / [f_1'(E_1) + f_2'(E_1)] \tag{22}$$

where we define

$$l_{21}(E_1) = -h_{21}(E_1) + [h_{21}'(E_1)] \times (1-E_1) \tag{23}$$

$$m_{12}(E_1) = h_{12}(E_1) + (h_{12}'(E_1)) \times (E_1) \tag{24}$$

so that $l_{21}(E_1)$ incorporates the externalities passed to M_2 by M_1, and $m_{12}(E_1)$ reflects the external effects yielded to M_1 by M_2.

Next, differentiate (17) with respect to time and solve for $\dot{\lambda}_1$, yielding

$$\dot{\lambda}_1 = \frac{\dot{E}_1\{C_1''+C_1''-\lambda_1 f_1''-[-2h_{21}'+h_{21}''(1-E_1)]-(\lambda_1)(f_2'')-[2h_{12}'+h_{12}''E_1]\}}{f_1'+f_2'}, \tag{25}$$

where, for economy in (25) and henceforth, we suppress the argument E_1 in the functions C_1', C_2', C_1'', C_2'', f_1', f_2', f_1'', f_2'', h_{12}', h_{12}'', h_{21}' and h_{21}'', while realizing, importantly, that these are all functions of E_1. Finally, to derive the steady-state condition for E_1, substitute (22) and (25) into (18) and set $\dot{E}_1 = 0$, which yields

$$V'(S) = \frac{\{-[-h_{21} + (h'_{21})(1-E_1) + h_{12} + (h'_{12})(E_1)] + C_1' + C_2' + \lambda_2\}(r+\alpha)}{f'_1 + f'_2}. \tag{26}$$

The steady-state condition for E_1 in (26) displays characteristics common to one control, one state variable optimal control problems. On the LHS appears the marginal value of an augmentation to the stock of the ecosystem service. In the denominator of the right-hand side, there appear MESP terms, while the numerator includes marginal cost terms, the social rate of discount, the natural rate of depreciation of the stock, and the multiplier associated with the BMP effort endowment share constraint. Interpretations of (26) with regard to these terms are intuitive. For example, as marginal costs increase, marginal products (MESP values) would need to increase as well for (26) to hold, all else equal; or, as marginal costs increase, the marginal value of an increase in the ES stock (LHS) would also need to increase for (26) to hold, *ceteris paribus*.

In addition, however, the numerator includes terms incorporating the cross-manager external effects that are at work (the externalities yielded by M_1 that affect M_2, and vice versa). These terms may possess a variety of signs and magnitudes given that (1) the externalities flow in both directions (for the simplest case of two land manager categories), (2) the net external effects may be either positive or negative, (3) the absolute values of the net external effects imposed on M_1 may either be greater or less than those imposed on M_2, and (4) the magnitude of external effects yielded by a land manager depends on the level of BMP that the manager implements. These terms complicate the steady-state conditions as well as the comparative statics. Further, the forgoing model fails to take into account the inherent uncertainty in the ecosystem dynamics. Thus, even the simplest dynamic optimization framework, while useful in terms of laying out some key concepts and equations, results in relationships that do not yield simple analytical solutions. Therefore, in the next section, we employ a dynamic optimization procedure that takes into account uncertainty and is better suited for attaining numerical solutions.

STOCHASTIC DYNAMIC PROGRAMMING: AN APPLICATION TO WATER QUALITY

Consider the following management options among which the PWES program manager (principal) can choose: (i) No payments to land managers to fund BMP (No Action), (ii) payments to M_1 to fund BMP, or (iii) payments to M_2 to fund BMP. The choice problem is somewhat different than in the dynamic model described above; there the manager chooses the optimal allocation of payments to make to both M_1 and M_2.

It is possible to focus on any one or a collection of aquatic ecosystem attributes that may be affected by the BMP funded by PWES. For this illustration, we consider the attribute of water clarity (measured in feet using a secchi disk) for three reasons. First, water clarity is often used as an indicator of various other attributes/characteristics of waterbodies, such as extent of algae growth, degree of eutrophication and levels of dissolved oxygen. Second, it is a characteristic that is easily observed, including by recreationists, unlike other measures/indicators such as nutrient (phosphorus, nitrogen) concentrations, actual water temperatures relative to normal/desirable temperatures, et cetera. Third, economic valuation studies demonstrate that, because water clarity is rather easily observed and affects the quantity and quality of particular water-based recreation activities, changes in clarity lead to measurable changes in net economic values derived from waterbodies (Eiswerth et al. 2008). We use economic valuation results from such analyses in our application.

Our state variable S is water clarity whose value in the next period is influenced in part by BMP implemented by M_1 and M_2 in the current period. These can affect S via a number of mechanisms, such as reductions in the rates of discharge of sediment, phosphorus and nitrogen from lands upon which the BMP is applied. We measure clarity on a scale from 0 to 10 feet (0 to 3.05 meters) using five ranges, each of which has a midpoint as follows:

Range 1: 0-2 feet (Midpoint, denoted MP = 1)
Range 2: 2-4 feet (MP = 3)
Range 3: 4-6 feet (MP = 5)
Range 4: 6-8 feet (MP = 7)
Range 5: 8-10 feet (MP = 9)

A feature inherent to this choice problem involves the substantial uncertainties related to the ways in which BMP affects water clarity. This uncertainty is created in part by appreciable variations across sites in factors such as slope and soil type, as well as uncertain conditions such as weather (e.g., the frequency and magnitudes of precipitation events) and a host of other environmental variables that may influence clarity in the periods following BMP implementation. Stochastic dynamic programming (SDP) offers one approach for incorporating uncertainty related to (1) the general effectiveness of BMPs, (2) differences in BMP effectiveness across land managers, (3) the influence of cross-manager external effects, and (4) how the state variable (water clarity) evolves as a function of precipitation and other weather and site-specific environmental factors.

Consider a planner's objective of maximizing the present value of a future stream of ecosystem services delivered by the water clarity attribute. Assume that the value of the ecosystem services per given area (e.g., per hectare or streamside linear foot), $V(S_t)$, does not vary over time and is affected only by the value of the clarity measure at time t (S_t). The planner's objective function is:

$$\sum_{t=0}^{T-1} \rho^t [V(S_t) - E(l_t)] + \rho^T F(S_T), \tag{27}$$

where V is the value of the ecosystem services delivered per unit of land in the watershed, S is the value of water clarity (state variable), E is the amount the principal spends on payments for ecosystem services (costs), l_t is the choice of which agent to pay for BMP services in period t, $F(S_T)$ is the value of the ecosystem services in end period T (salvage value), and $\rho = 1/(1+r)$, where r = social rate of discount.

The general form of the equation of motion for S_t is:

$$S_{t+1} = g(S_t, l_t) + \varepsilon_t, \tag{28}$$

where ε is a random variable with zero mean and variance σ^2, and the initial condition is $S_0 = \bar{S}$. Equation (28) is the Markov condition: the current level of water clarity is a function solely of last period's clarity level and the principal's decision regarding which agent (land manager) to pay in the current period. The evolution of clarity over time thus depends on the level of S and which agent is paid in period t, and stochastic processes given by ε. The Bellman recursive equation for the SDP problem is:

$$W_t(S_t, l_t) = \max_{l_0, l_1, \ldots, l_{T-1}} \{\mathrm{E}[V(S_t) - E(l_t)] + \rho \sum_{j=1}^{k} P(i, j, l_t)\, W_{t+1}[S_{t+1}(j)]\} \tag{29}$$

where $P(i,j,l_t)$ represents the probability that $S_t(i)$ (i=1,…,n) will transition to state $S_{t+1}(j)$ (j=1,…,n), given that PWES option l (l=1,…,L) is chosen in period t. W_t represents the expected discounted value of the future stream of net ecosystem service values in period t, given the level S in period t and assuming that the optimal path is taken in every future period.

We develop a set of transition probability matrices (TPMs) indicating the expected transition of the state variable S (water clarity) for various combinations of (1) the planner's choice of action and (2) the nature of the cross-manager external effects yielded by the BMPs implemented by M_1 and M_2. Because there are three possible actions the principal can take in this illustration, there are at least 16 possible combinations of M_1 and M_2 external effects, since the effects may be either positive or negative for either agent type and the absolute values of the externalities may be labeled either 'relatively high' or 'relatively low' (at the simplest level) when comparing M_1 and M_2. For example, M_1 may yield high positive externalities relative to M_2's low positive externalities; M_1 may yield high negative externalities relative to M_2's low positive externalities; and so on for sixteen combinations (assuming external effects are nonzero for both land managers). We consider a small set of possibilities for straightforward illustration because it is neither necessary nor useful to consider every possible combination.

Transition Probability Matrices and Parameter Values for the SDP Model

Transition Probability Matrices

In Table 1, we illustrate how the transition matrices might be expected to differ from one another for the following five selected cases:

- Case 0: 'No PWES' (No Payments/Action) [NP];
- Case 1A: Make Payments to M_1, Assuming No Externalities [Pay M_1, No EXT];
- Case 1B: Make Payments to M_1, Assuming M_1 Yields High Positive Externalities Relative to M_2 [Pay M_1, High Pos M_1 EXT];
- Case 2A: Make Payments to M_2, Assuming No Externalities [Pay M_2, No EXT]; and
- Case 2B: Make Payments to M_2, Assuming M_1 Yields High Positive Externalities Relative to M_2 [Pay M_2, High Pos M_1 EXT].

Note that, with the five cases above, we do not actually have five controls. Rather, we have three controls: (1) Do Not Pay, (2) Pay M_1, and (3) Pay M_2. Different SDP scenarios are then conducted for alternative assumptions about the nature of the externalities at work, but always with three controls. In our illustration, we consider the differences between a case of no externalities and one of relatively high positive externalities yielded by M_1.

In developing the transition probability matrices shown in Table 1, we begin by positing reasonable values for the “No Payments/Action” [NP] matrix.[2] Then, we develop elements for the *Pay M_1, No Externalities* matrix as follows. Assume that, if the PWES program chooses to fund BMPs in a particular period, it is able to fund eight farms. Assume that there are 50,000 acres of farmland in the watershed, and a total of 200 farms with a mean farm size of 250 acres.[3] If the PWES program can fund eight farms per year, this implies funding BMPs on 2,000 acres/yr. Therefore, it would take 25 years to fund BMPs on all of the farmland acres in the watershed, with 4% of the farmland acres funded in each period.

Next, assume that BMP nutrient reduction efficiency of the funded BMPs equals 25%. For reference, Shortle et al. (2013, p.28) summarize data collected from various sources regarding the nitrogen, phosphorus and sediment reduction efficiencies for eighteen different crop or livestock management BMPs. For example, the BMP for cover crops is estimated to have reduction efficiencies for nitrogen, phosphorus and sediment of 34%, 0-15%, and 0-20%, respectively. While reduction efficiencies may vary widely across BMP, crops, and farm locations, our use of an assumed 25% nutrient and sediment reduction efficiency would be in line with estimates for such a BMP or similar agricultural practices.

[2] Our application of SDP is to an illustrative watershed rather than a specific watershed with site-specific data from scientific field studies and models. Nevertheless, to solve the model we do utilize data collected by studies from various watersheds in the U.S. Our intent is to illustrate generally how the approach may be applied to an actual watershed, which would be one logical next step in this line of research.

[3] For comparison, the average number of acres harvested per farm is 260 acres in Weld County, Colorado’s largest agricultural county, which is located in the South Platte River Basin (calculated from 2012 Census of Agriculture, Volume 1, Chapter 2, Table 9). In Wisconsin, home to the waterbody for which net economic values for water clarity are estimated in Eiswerth et al. (2008) and used in this manuscript, mean acres per farm are approximately 210 acres (calculated from 2012 Census of Agriculture, Volume 1, Chapter 1, Table 11).

Table 1. Transition Probability Matrices for Various Cases/Control Variables[a]

Control and Current State of WC (Midpoint in feet)	Future State of WC (Midpoint of Range, in feet) 1	3	5	7	9
Case 0 (control 0): NP Matrix					
1	0.95	0.03	0.02	0	0
3	0.24	0.70	0.03	0.02	0.01
5	0.16	0.19	0.60	0.03	0.02
7	0.05	0.07	0.15	0.70	0.03
9	0.01	0.04	0.10	0.15	0.70
Case 1A (control 1): Matrix for *Pay M_1, No EXT*					
1	0.91	0.05	0.03	0.01	0
3	0.21	0.66	0.07	0.04	0.02
5	0.08	0.18	0.66	0.05	0.03
7	0.04	0.09	0.14	0.66	0.07
9	0.005	0.025	0.07	0.12	0.78
Case 1B (control 1): Matrix for *Pay M_1, High Pos. M_1 EXT*					
1	0.80	0.08	0.06	0.05	0.01
3	0.20	0.60	0.10	0.07	0.03
5	0.08	0.10	0.60	0.13	0.09
7	0.02	0.03	0.10	0.65	0.20
9	0.005	0.015	0.03	0.07	0.88
Case 2A (control 2): Matrix for *Pay M_2, No EXT*					
1	0.88	0.06	0.04	0.02	0.00
3	0.24	0.64	0.07	0.04	0.01
5	0.09	0.15	0.64	0.07	0.05
7	0.02	0.07	0.12	0.67	0.12
9	0.01	0.02	0.04	0.10	0.83
Case 2B (control 2): Matrix for *Pay M_2, High Pos. M_1 EXT*					
1	0.88	0.06	0.04	0.02	0.00
3	0.24	0.64	0.07	0.04	0.01
5	0.09	0.15	0.64	0.07	0.05
7	0.02	0.07	0.12	0.67	0.12
9	0.01	0.02	0.04	0.10	0.83

[a] Row sums are equal to 1.0. Note that the Case 2B matrix (Pay M_2, High Positive M_1 Externalities) is identical to the Case 2A matrix, because the fact that M_1 yields positive externalities if funded does not matter in the event that M_2 is funded instead.

With the above estimates in place, we assume that, if 4% of the farms (acres) are treated each year, there will be a watershed-wide nutrient reduction efficiency of 0.01 0.04×0.25), or 1% assuming linearity in effects. To translate this into probabilities for changes between clarity states, one of the simplest possible assumptions would be that the baseline probability *P* of transitioning from the current level of water clarity to the next higher level (as shown in the Case 0 transition matrix) will increase by 0.04 (denoted as ΔP) and that the other probabilities in the row remain the same, except for the probability of remaining at the current clarity level (which decreases) and the probabilities of transitioning to lower levels of clarity (which also decrease). The Case 1A transition matrix in Table 1 is developed through application of these principles.

Next, the Case 1B matrix (for "Pay M_1, High Positive M_1 Externalities") is developed as follows. Assume that, if M_1 yields high positive externalities, the four farms immediately adjacent to each funded farm (land manager) implement (as a result of social diffusion, say) the same BMP. Then, the impact of the funding will increase by a factor of five. That is, for each farm funded by the PWES program, the program now gets five farms that adopt the BMP instead of just one. If the PWES program funds eight farms/yr as discussed above and as a result induces implementation of BMPs on 40 farms/yr, then that represents 40/200 or 20% of the 200 farms in the watershed. Thus there is now a watershed-wide nutrient reduction efficiency of 0.05 (=0.20×0.25), or 5%, within any period (year), as compared to 1% under no externalities (Case 1A). The Case 1B matrix in Table 1 is developed by adjusting the baseline (Case 0) transition probability matrix according to these principles, that is, similar to the derivation of the Case 1A matrix but accounting for the 'multiplier effect' induced through the positive externalities of social diffusion.

Finally, the Case 2A and 2B matrices represent "Pay M_2, No Externalities" and "Pay M_2, High Positive M_1 Externalities." These are developed as follows: We assume that the implementation of BMP by managers of type M_2 is expected to result in higher levels of nutrient reduction, and hence larger increases in clarity (the state variable), than BMP implementation by managers of type M_1. Specifically, assume that funding M_2 is twice as effective as funding M_1. This implies that, when one M_2 acre is funded for BMP, the edge-of-field nutrient reduction efficiency thereby attained is equivalent to funding roughly two M_1-type acres. If each year the principal funds eight farms (as assumed throughout) of type M_2, that will then yield the same nutrient reduction results as funding 16 type-M_1 farms. Therefore, the watershed-wide nutrient reduction efficiency will be twice the value as in Case 1A, or 0.02 (=0.08×0.25). Employment of these assumptions results in the transition probability matrix for Case 2A. Lastly, the Case 2B matrix (Pay M_2, High Positive M_1 Externalities) is identical to the Case 2A matrix, since the fact that M_1 yields positive externalities if funded does not matter in the event that M_2 is funded instead in a given period.

Values for the BMP Cost Parameter

Values for the parameter $E(l_t)$, the expenditure on BMP, are obtained from Shortle et al. (2013, p.16), who report BMP unit costs both by specific BMP and by various states in the Chesapeake Bay watershed.[4] For example, for the BMP we reference above with regard to nutrient reduction efficiencies (cover crops), estimated unit costs are reported for six states and range from \$40/acre per year in Pennsylvania to \$109/acre in Virginia. In our model, we assume somewhat higher BMP unit costs for M_2 in the same way that we assume higher nutrient reduction efficiencies for M_2, as described above. Specifically, we use the following as baseline parameter values for unit costs:[5]

$$\text{Baseline BMP unit cost for } M_1 = \$50/\text{acre/yr} \tag{30}$$

[4] To develop estimates of BMP unit costs, Shortle et al. (2013) utilize data from Abt Associates/ USEPA (2012) and make some modifications.

[5] For comparison, the mean of BMP unit costs for cover cropping for the three lowest-cost states (Pennsylvania, Delaware, Maryland) is approximately \$53/acre per year, while it is roughly \$94/ac for the 3 highest-cost states (New York, West Virginia, Virginia) (Shortle et al. 2013, p.16).

Baseline BMP unit cost for M_2 = \$80/acre/yr (31)

Therefore, if the PWES program funds BMP on eight farms per year for an annual total of 2,000 acres, as assumed above, then the baseline values for total annual cost $E(l_t)$ are:

Baseline $E(l_t)_{M1}$ = 2,000 acres/yr × \$50/acre/yr = \$100,000/yr (32)

Baseline $E(l_t)_{M2}$ = 2,000 acres/yr × \$80/acre/yr = \$160,000/yr (33)

The influence of changes in the values of $E(l_t)$ for each land manager type may be examined via sensitivity analyses.

Values for the Benefits Function $V(S_t)$

Values for the function $V(S_t)$ are from Eiswerth et al. (2008), who estimated consumer surplus values from recreational angling as a function of different levels of water clarity at a Wisconsin lake. Specifically, they estimated a consumer surplus value of \$1.38 million/yr at 10 ft clarity and that an actual historical decline in water clarity from 10 ft to 3 ft resulted in an approximate 37% decrease in aggregate consumer surplus values. Assuming linearity in $V(S_t)$ over the range from 1 ft to 10 ft, we estimate consumer surplus values as a function of clarity, $V(S_t)$, as shown in Table 2.

Table 2: Consumer surplus values as a function of water clarity ($V(S_t)$).[a]

Water Clarity (feet): S_t	Annual Consumer Surplus (millions of \$/yr): $V(S_t)$
1	0.708
3	0.858
5	1.007
7	1.156
9	1.305

[a] Values derived from Eiswerth et al. (2008).

SDP Model Results

The SDP outcomes are provided in Table 3, which presents the optimal PWES strategies for various starting values of the state variable. The results in column 2 are for the case where no cross-manager externalities are at work, while those in column 3 are associated with the case of positive externalities from BMP adopted by M_1 as detailed in the previous section. Focusing first on the case of no externalities, the results indicate that, if water clarity is quite low (=1 or 3), the optimal strategy involves no payments by the principal to either agent. The intuition here is that, starting from very low states of water quality, the payback is not high enough to justify expenditures. As baseline water clarity rises to an intermediate level of quality (=5), however, the optimal strategy shifts from *No Pay* to *Pay* M_1. Recall that M_1 is assumed to have lower BMP unit costs than M_2. The intuition is that the improved water quality purchased by paying the lower-cost provider of ecosystem services is now justified due to the higher baseline water quality level, and hence higher ending level of water quality attained following PWES payments. Further, if baseline water quality is even higher (=7 or 9), the optimal strategy shifts again from *Pay* M_1 to *Pay* M_2. The intuition for the optimal strategy under these higher levels of baseline quality may be expressed as follows: given the payback from maintaining water clarity at highly desirable levels, the PWES manager now finds it optimal to make payments to the land manager type M_2 that, while having higher BMP unit costs, is also more effective implementing BMP, as indicated by a higher edge-of-field nutrient reduction efficiency.

Turning to column 3, we find that, in a context where there are positive externalities from M_1's BMP, the optimal strategy involves *Pay* M_1 regardless of the baseline state of water quality. This represents a substantial change in strategy relative to a case where externalities are not at work or, equivalently, the PWES manager assumes incorrectly that no externalities exist. As shown in Table 3, the scenario of positive externalities from M_1's BMP also results in optimal strategies yielding higher overall annual expected returns from PWES, as compared to the no-externalities context.

It is possible to use the SDP model to determine optimal strategies for a large number of combinations among (1) alternative assumptions regarding the signs and magnitudes of cross-manager externalities, and (2) values of key model parameters. Due to space constraints, we limit the presentation of results to those shown in Table 3; we discuss some of the more general implications in the concluding section.

Table 3: SDP Results: Optimal PWES Strategies.

Clarity States (feet)	Context/Scenario	
	Context A: No Externalities	Context B: Positive Externalities from M_1 BMPs
1	No Pay	Pay M_1
3	No Pay	Pay M_1
5	Pay M_1	Pay M_1
7	Pay M_2	Pay M_1
9	Pay M_2	Pay M_1
Overall Annual Expected Returns	$767,588	$994,124
SD of Annual Returns	$111,697	$228,000

[a] Baseline parameters: $V(S_t)$ at 10 ft clarity = $1.38 million/yr [=1×Delavan Lake angling consumer surplus]; discount rate = 5%; T=60 yrs.

CONCLUSIONS

We identified the issue of cross-manager externalities as a potentially important topic in PWES design and implementation, and examined the implications of accounting for externalities using three analytical frameworks. First, to shed initial intuitive light on the topic, we introduced the problem from a static perspective. Then, to develop insights on the dynamic nature of the problem, we formulated the PWES planner's decision using an optimal control framework. Finally, to determine optimal strategies for the PWES planner, we employed a stochastic dynamic programming approach. Our methods contribute to the literature by allowing for identification of dynamically optimal PWES payment patterns, whether externalities exist or not. This is useful because, to date, such analyses have not been conducted to our knowledge. Importantly, our approach also shows how optimal payment strategies change when the principal is able to recognize, estimate and account for BMP externalities. This is important because such externalities are present in many situations.

In addition, recent literature suggests that, despite the fact that some researchers have argued against it, practitioners should in fact consider allowing PWES programs to recognize side objectives. Such side objectives (spillover impacts) may include social, economic or community benefits that arise from the institutions, processes or social relationships that develop when PWES payments are made to land managers. These are benefits not strictly related to those stemming from enhanced ecosystem services. Though the cross-manager externalities considered here were described as the effect that one land manager's PWES-funded BMP may have on another manager's ability to enhance ecosystem services using a BMP, our models certainly could be applied to a wider spectrum of PWES externalities and benefits. Thus, our study may enhance other researchers' efforts to account for community or economic externalities resulting from PWES expenditures, as well as assist PWES managers to more fully and accurately consider such side impacts in their payment allocation decisions.

There are four priority areas in which we believe researchers should focus next steps. First, high value-added is expected from research to collect monitoring or expert opinion data to inform the types of state variable transition probabilities used in our SDP model. Second, researchers should consider implementing primary surveys with land managers to uncover and quantify patterns of historical or expected social diffusion and other spillovers in response to PES. Such research would also feed into the first need in terms of increasing the accuracy of transition probability matrices when externalities are predicted. Third, the next iteration of our approach could move from illustration to application.

Fourth, economists have performed substantial research for many years in other related venues – these efforts have yielded germane methods and results that have not yet been adequately transferred to the arena of PWES and PES. For example, economics research in areas such as natural resource damage assessment and regulatory benefits estimation has resulted in both methodological advancements (e.g., in nonmarket valuation) and results (e.g., monetary values associated with changes in water quality at a site) that can be more fully exported to the realm of PWES. Transfer of both benefit estimates (benefit transfer) and valuation methods is a highly cost-effective approach for increasing the efficiency of PWES funds allocation, but to our knowledge is an approach that remains largely untapped to date.

REFERENCES

Abt Associates. 2012. Unit Costs of Agricultural Best Management Practices (BMPs) in Watershed Implementation Plans (WIPs) for the Chesapeake Bay Jurisdictions. Prepared for U.S. Environmental Protection Agency. Cambridge, MA. July 2012.

Bennett, G., and N. Carroll. 2014. Gaining Depth: State of Watershed Investment 2014. Available online at www.ecosystemmarketplace.com/reports/sowi2014.

Brown, T.C., J.C. Bergstrom, and J.B. Loomis. 2007. Defining, valuing, and providing ecosystem goods and services. *Natural Resources Journal* 47(2): 329–376.

Buttoud, G. (editor), 2012. Emerging Economic Mechanisms for Global Forest Governance, *Forest Policy and Economics* (Special Issue) 18(1): 1-52.

Daily, G. C. 1997. *Introduction: What Are Ecosystem Services?* In: Daily, G.C. et al. (eds.), Nature's Services: Societal Dependence on Natural Ecosystems 1, 3.

Daniels, A.E., K. Bagstad, V. Esposito, A. Moulaert, and C.M. Rodriguez. 2010. Understanding the impacts of Costa Rica's PES: are we asking the right questions? *Ecological Economics* 69: 2116–2126.

Eiswerth, M.E., R. Kashian, and M. Skidmore. 2008. "Examining Angler Behavior Using Contingent Behavior Modeling: A Case Study of Water Quality Change at a Wisconsin Lake." *Water Resources Research* 44, W11426, doi:10.1029/2006WR005828. Available at: http://www.agu.org/journals/wr/wr0811/2006WR005828/2006WR005828.pdf

Engel, S., S. Pagiola, and S. Wunder. 2008. Designing payments for environmental services in theory and practice: an overview of the issues. *Ecological Economics* 65: 663–674.

Farley, J., and R. Costanza. 2010. Payments for ecosystem services: from local to global. *Ecological Economics* 69 (11): 2060–2068.

Fisher, B., Turner, K., Zylstra, M., Brouwer, R., Groot, R.D., Farber, S., Ferrano, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D., and Balmford, A. 2008. Ecosystem Services and Economic Theory: Integration for Policy-Relevant Research. *Ecological Applications* 18: 2050–2067.

Goldman-Benner, R.L., S. Benitez, T. Boucher, A. Calvache, G. Daily, P. Kareiva, T. Kroeger, and A. Ramos. 2012. Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. *Oryx* 46(1): 55-63.

Kemkes, R.J., J. Farley, and C.J. Koliba. 2010. Determining when payments are an effective policy approach to ecosystem service provision. *Ecological Economics* 69: 2069-2074.

Kolinjivadi, V., J. Adamowski, and N. Kosoy. 2014. Recasting payments for ecosystem services (PES) in water resource management: A novel institutional approach. *Ecosystem Services* 10: 144-154.

Lin, H. 2012. The global science of integrated water governance. *Journal of Environmental Science and Engineering* B 1: 1167-1174.

Lin, H., J.A. Thornton, and T.M. Slawski. 2013. Participatory and evolutionary integrated lake basin management. *Lakes & Reservoirs: Research and Management* 18: 81–87.

Malmsheimer, R.W., J.L. Bowyer, J.S. Fried, E. Gee, R.L. Izlar, R.A. Miner, I.A. Munn, E. Oneil and W.C. Stewart, 2011. Managing Forests because Carbon Matters: Integrating Energy, Products, and Land Management Policy, *Journal of Forestry* 109(7S): 7-50.

Millennium Ecosystem Assessment. 2003. Ecosystems and Human Well-Being: A Framework for Assessment. Washington, D.C.: Island Press.

Norgaard, R. 2010. Ecosystem services: from eye-opening metaphor to complexity blinder. *Ecological Economics* 69(6): 1219–1227.

Shortle, J., Z. Kaufman, D. Abler, J. Harper, J. Hamlett, and M. Royer. 2013. *Building Capacity to Analyze the Economic Impacts of Nutrient Trading and Other Policy Approaches for Reducing Agriculture's Nutrient Discharge into the Chesapeake Bay Watershed*. Final Report prepared for the USDA Office of the Chief Economist, Cooperative Agreement No. 58-0111-11-006, August 2013.

Sommerville, M.M., J.P.G. Jones, and E.J. Milner-Gulland. 2009. A revised conceptual framework for payments for environmental services. *Ecology and Society* 14: 34–47.

Wunschler, T., S. Engel, and S. Wunder. 2008. Spatial targeting of payments for environmental services: a tool for boosting conservation benefits. *Ecological Economics* 65: 822-833.