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METHODS

Multiple attribute decision analysis for ecosystem management

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Abstract

Implementation of an ecosystem approach to natural resource management requires evaluation of a broad array of ecological services in a multidimensional, community-based watershed approach that empowers people to make informed management decisions. Conventional economic approaches that assign values to ecological services (contingent valuation) or that evaluate the efficiency of preserving and restoring those services (cost–benefit analysis) are quite limited for this purpose. In addition to the methodological difficulties encountered in applying contingent valuation and cost–benefit analysis, several problems occur when nonmarket values of ecological services are estimated independently of ecosystem planning and management. Multiple attribute decision-making (MADM) is an alternative conceptual framework for evaluating and selecting land and water resource management systems (LWRMS). Advantages of MADM are that it facilitates community-based collaborative decision-making, avoids some of the ethical, theoretical and practical shortcomings of conventional economic approaches, does not require assigning monetary values to ecological services, allows consideration of multiple attributes and is not culturally biased. The MADM model described in this paper explains how a property manager selects the most preferred LWRMS for a property based on their multiple stochastic attributes. Application of the model requires determination of the technically feasible LWRMS for a property and specification of the socially acceptable ranges of attributes. This information is combined with economic/biophysical simulations to derive the efficient combination of attributes and LWRMS for a property. The property manager then selects the most preferred combination of attributes from the efficient combinations of attributes for a property using utility maximization, surrogate worth tradeoff, free iterative search, analytical hierarchy process, Aspiration–Reservation Based Decision Support System or stochastic dominance. A watershed alliance can evaluate the sustainability of the most preferred LWRMS for properties in a

Abbreviations: ARBDSS, aspiration–reservation based decision support system; CBA, cost–benefit analysis; CV, contingent valuation; EM, ecosystem management; LWRMS, land and water resource management systems; MADM, multiple attribute decision making; NID, normally independently distributed; WTP, willingness to pay.

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watershed based on a weak or strong sustainability criterion. If the alliance determines that the most preferred LWRMS are not sustainable, then an index of attributes is used to evaluate the cost effectiveness of alternative public policies for stimulating the adoption of more sustainable LWRMS.

Keywords: Ecosystem management; Contingent valuation; Cost–benefit analysis; Multiple attribute decision making

1. Introduction

Ecologically sustainable management of natural resources, also known as ecosystem management (EM), requires a complete and accurate accounting of the social, economic and ecological impacts of human activities. Concern with ecological impacts of human activities has increased in importance due to the shift in resource management philosophy from production of food, fiber and forage to protection and restoration of ecosystems (National Research Council, 1992; Williams et al., 1997). EM incorporates larger spatial scales, longer time periods and more variables than commodity-based resource management (Thomas, 1997) and focuses on achieving and sustaining a balance between human and ecological values (MacKenzie, 1996). Many ecological services, such as air and water purification, mitigation of floods and drought, detoxification and decomposition of wastes, generation and renewal of soil, maintenance of biodiversity and partial stabilization of climate are not valued in the marketplace (Daily, 1997).

The objectives of this paper are to (1) evaluate the weaknesses of nonmarket valuation and cost–benefit analysis (CBA) and the strengths of multiple attribute decision making (MADM) in evaluating land and water resource management decisions; (2) present a conceptual framework for implementing MADM in property management and evaluating sustainable resource management in watersheds and (3) develop a method for assessing policies designed to enhance ecologically sustainable resource management. It is argued that MADM overcomes several weaknesses of nonmarket valuation and CBA in evaluating the ecological consequences of resource management decisions.

2. Nonmarket valuation

Ecosystems cannot be managed sustainably without first understanding how the services they provide are impacted by human activities. Risser (1990) notes, "…there has been an increasing recognition that prudent management of the world's natural resources requires that human-defined parts of the landscape must be managed in the context of the interactions with the more natural parts of the landscape" (p. 46). Ecological assessment requires an understanding of how ecological services influence and are influenced by land and water resource management. Postel and Carpenter (1997) claim that, "Better accounting for nonmarket values of rivers, lakes and wetlands would help ensure that land-use and water management decisions are both economically rational and environmentally sound" (p. 210). The most common way that economists account for the ecological impacts of human activities is to estimate the monetary value of reducing those impacts or preventing them from occurring using surrogate market techniques, such as travel cost and hedonic pricing, and nonmarket valuation methods, such as contingent valuation (Prato, 1998b).

Kahn (1996) criticizes nonmarket valuation methods, and contingent valuation (CV) in particular, on several grounds. First, CV is a single-attribute valuation technique that is poorly suited for evaluating the multifaceted ecological impacts of resource management decisions. Conjoint analysis, which is a multiple attribute extension to CV, is not subject to this criticism. Second, many non-economists reject the assignment of monetary values to ecological services based on ethical considerations. Third, some economists argue that willingness to pay (WTP) measures are likely to be biased by imperfect information on the part of the respondent, embedding of the value of other goods in stated willingness to pay values and other response biases. In summarizing CV's weaknesses, Kahn (1996) indicates that contingent valuation "…is associated with controversy and is far from universally accepted, even among environmental economists" (p. 8). Bjornstad and Kahn (1996), Cummings (1996), Mitchell and Carson (1989) and Smith (1992) also discuss the strengths and weaknesses of CV.

Another criticism of CV is that survey respondents tend to express their WTP or willingness to accept compensation for a good or service from the viewpoint of a concerned citizen rather than as a consumer or user of that good or service (Sagoff, 1988). Cameron (1997) commented that respondents in a survey in the Hawkesbury–Nepean watershed (Sydney, Australia) found it difficult to state their WTP for improved water quality. Other respondents stated that the payments they would be willing to make are symbolic and like a donation to a worthy cause.

Estimation of nonmarket values of ecological services is done, for the most part, independently of ecosystem planning and management. This causes several problems. First, monetary estimates of ecological services are likely to be ignored by resource planning/management groups that are critical of nonmarket valuation methods. Second, there is a greater likelihood of misinterpretation or misapplication of nonmarket values when estimation is done independently of application. Third, since nonmarket values of ecological services are problem- and area-specific, ecosystem values derived for one area are generally not suitable for other areas.

Estimating and applying monetary values of ecological services in the same study is empirically burdensome because of the diversity and complexity of those services. Not only does this burden make it more difficult to account for ecological impacts but it also increases the likelihood that resource management decisions are biased in favor of economic-based as opposed to ecologically-based considerations (Goulder and Kennedy, 1997). A classic example of this pitfall is the development of hydropower in the Columbia River system in the Pacific Northwest. Emphasis on economic development benefits and lack of understanding and evaluation of ecological impacts led to development of a hydropower system that drastically reduced salmon populations and associated socioeconomic and cultural benefits from their pre-European settlement levels (Lee, 1995).

3. Cost–benefit analysis

Aggregate WTP estimates of ecological services estimated with CV can be incorporated in CBA of resource investments (Feather et al., 1995; Cameron, 1997). Even if CV methods had none of the above limitations, incorporating monetary values of ecological services in CBA is problematic. CBA requires discounting all benefits and costs of investments that occur over a given planning horizon. Discounting cash flows from ecological investments that have long-term and uncertain benefits and high and short-term costs reduces the net present value of those investments and results in under-investment in ecological protection or restoration. This problem arises, for example, when evaluating policies to reduce greenhouse gas emissions and their potentially adverse effects on global climate.

Goulder and Kennedy (1997) give additional reasons why CBA is not a sufficient criterion for evaluating natural resource investments. First, it does not consider the intra-generational and inter-generational fairness of the distribution of gains and losses because it is primarily an economic efficiency criterion. Second, basing ecological investment decisions on CBA does not ensure sustainability in terms of maintaining essential ecological services. Fairness and sustainability limitations of CBA are widely recognized (Bishop, 1993; Perrings, 1994). Third, using a point estimate of ecological impacts in CBA ignores the uncertainty regarding the ecological impacts of human activities.

Joubert et al. (1997) argue that CBA is not an appropriate tool for evaluating investments that generate social and environmental externalities. They elucidate several advantages of MADM relative to CBA. First, MADM facilitates public participation, allows comparisons of investments based on stakeholders' preferences and is well suited for collaborative decision-making (Yaffee and Wondolleck, 1997) and scientific assessments (Johnson, 1997). In contrast, CBA is a top-down approach that is characteristic of rational comprehensive planning for natural resource management (Smith, 1997).

Second, MADM circumvents the classical utilitarian view of optimality in decision making by avoiding some of the ethical, theoretical and practical shortcomings of CBA. Third, MADM does not require assigning monetary values to ecological services. Fourth, CBA reduces the multiplicity of criteria and objectives underlying most decisions to a single monetary criterion (net present value). MADM allows attributes to be evaluated separately and makes the evaluation more accessible to decision-makers. Fifth, the underlying assumptions of CBA are less appropriate in developing than developed countries because residents of many developing countries operate outside of any formal market setting. Since MADM only requires enumeration of preferences for attributes, it is adaptable to illiterate people. The specific approach taken here is not designed for illiterate people. In describing the foundations of socio-ecological economics, Jacobs (1966) states axiomatically that, "individual economic behavior is determined by social structure and institutions, culture and ethical values, as well as by (in some circumstances) utility maximization" (p. 15). This axiom is compatible with MADM.

4. Watershed approach

One of the best ways to implement EM is through a community-based watershed approach that empowers people to make informed management decisions. A top-down approach is unappealing to landowners and rural communities because it generally provides results and recommendations that lack practical significance and broad-based community support. Lee and Stankey (1992) point out that, "Large-scale (regional) ecological systems can be most effectively regulated by small-scale (local) social organizations" (p. 247). Naiman et al. (1997) indicate that "...watershed management demands unparalleled cooperation between citizens, industry, governmental agencies, private institutions, and academic organizations" (p. 251). Local social organizations and cooperation require decentralized decision making. CBA is a top-down evaluation technique that is less compatible with decentralized, community-based decision making than MADM (Cameron, 1997).

A watershed approach is supported by the Clean Water Act, National Water Agenda for the 21st Century (Water Environment Foundation, 1992) and many governmental agencies and professional organizations (Adler et al., 1993; USEPA, 1994). Central elements of a watershed approach are ecological health, strong communities, sustainable resource use, clean air and water and natural resource-based activities (Durning, 1996). A watershed approach recognizes that people depend on and must assume responsibility for the ecosystems in which they live (National Wildlife Federation, 1997).

In many areas of the world, watershed alliances have formed to evaluate the cumulative ecological effects of land use and water resource management in watersheds. Alliances typically include a wide range of stakeholders such as property managers, federal and state resource management agencies, commodity and environmental groups, local government, private industry and others. The basic premise underlying the formation of watershed alliances is that assessments of sustainable resource management and the design of policies to alleviate unsustainable resource management should occur at the local level. A watershed alliance can utilize MADM to evaluate the social, economic and ecological sustainability of resource management. If a watershed alliance or environmental authority determines that resource management is not sustainable, then it is appropriate for them to evaluate alternative policies (education, technical assistance and economic incentives) that encourage sustainable resource management.

Land ownership has important consequences for applying MADM. Private land managers select land and water resource management systems

(LWRMS) for their properties based on natural and anthropogenic factors including personal motivations and preferences; social, cultural, environmental and economic conditions; and public policies. LWRMS refer to the spatial pattern of land and water resource management practices for a property or watershed. A MADM approach is well suited for selecting the most preferred LWRMS for private property because there is typically only one property manager.

LWRMS for public land are selected by a federal or state agency. MADM is more difficult to apply to public land than private land because public land management must consider a wide range of private and public interests and involves many stakeholders with potentially conflicting interests. Classic examples of the latter include the management of national forest lands in the Pacific Northwest to protect habitat of the northern spotted owl and the killing of bison that enter private lands adjacent to Yellowstone National Park. Management of public lands requires careful consideration of how resource management decisions affect different stakeholders.

5. Nature of ecosystem management

This section explores the nature of EM within the context of MADM. EM represents a fundamental shift in the philosophy of managing land and water resources. While EM is applicable to both public (state and federal) and private lands, public land managers and environmental groups are the strongest proponents. Eighteen federal agencies have adopted or are considering adoption of programs based on an ecosystem approach to land and water resource management (Haeuber and Franklin, 1996).

Diaz and Bell (1997) point out that, "…on federal lands [in the United States] the concept of resource management (in the sense of managing the production of individual resources like timber, minerals, forage for livestock, and scenery) has virtually given way to the more systematic view of ecosystem management—managing the patterns and processes in a holistic manner to provide for sustained character and function, as well as for benefits and commodities for humans" (p. 256). Thomas (1997) maintains that, "…ecosystem management is only a concept for dealing with larger spatial scales, longer time frames, and many more variables (ecological, economic, and social) than have commonly been considered in past management approaches" (p. xi). Successful EM attempts to manage for sustainable productivity of the whole ecosystem (Schowalter et al., 1997).

Implementation of EM is challenging because its larger spatial scales and longer planning horizons increase biophysical and organizational complexity. Biophysical complexity increases the uncertainty regarding how LWRMS influence ecological integrity. Organizational complexity necessitates the creation of "organizational forms that can simultaneously coordinate scientific inquiry and democratic deliberation with existing institutions for managing natural resources…" (p. 362) and a shift of policy-making activity toward the local level (Meidinger, 1997).

One approach for dealing with biological uncertainty is adaptive resource management (Holling, 1978; Walters, 1996). Its basic premise is that, "if human understanding of nature is imperfect, then human interactions with nature [e.g. policies] should be experimental" (Lee, 1995, p. 229). While adaptive management is an appropriate response to biological complexity, it is time consuming and expensive and can give faulty results when relevant variables are either ignored or not held constant (Smith, 1997). Despite these weaknesses, experiments provide useful information to stakeholders and policy makers.

Another way to handle biological uncertainty is through the application of computer simulation models. Computer models are one of the best methods of evaluating the broad range of space and time scales that characterize EM (Gunderson et al., 1995). Not all computer models are appropriate for EM. Behan (1997) points out that, "Virtually all of the simulation models to date are science-based, quantitative, linear programming optimizing applications. This represents a terrible constraint—one that biosocial forestry cannot tolerate" (p. 416). Linear programming models are frequently applied the same way as CBA, namely, in a top-down, technocratic manner.

EM might best be implemented by combining results from simulation analysis of LWRMS with adaptive management. In this approach, simulation models are used to evaluate the biophysical, economic and ecological consequences of LWRMS and adaptive management is used to monitor and evaluate the impacts of different LWRMS and public policies. Knowledge gained from adaptive management is useful for informing stakeholders, policy makers and others about the socioeconomic and ecological impacts of specific LWRMS and public policies and to refine and validate the simulation models.

6. MADM model

This section discusses a MADM model that describes how a property manager selects the most preferred LWRMS for a property. The basic premise underlying MADM is that a property manager selects LWRMS for a property based on their multiple and stochastic attributes. The most preferred LWRMS for a property provide the greatest achievement of the attributes. MADM has been used or proposed for water resources systems analysis (Haimes and Hall, 1974), environmental management (Janssen, 1992), food security (Haettenschwiler, 1994), forest management (Penttinen, 1994), farm-environmental management (Xu et al., 1995), regional water quality analysis (Makowski et al., 1995), management of agroecosystems (Prato et al., 1996a), wildlife management (Prato et al., 1996b) and soil and water resource management (Prato, 1998c). Unlike CBA, MADM does not require the estimation of monetary values for non-market ecological services provided by LWRMS such as flood protection, water purification and biodiversity.

The MADM model proposed here can be used to identify the most preferred LWRMS for properties in a watershed. For example, Hajkowicz and Prato (1998) used a non-stochastic, utility-based MADM model as a basis for developing a farm survey. Results from the survey were used to determine farmers' preferences for attributes and alternative farming systems in Missouri's Goodwater Creek watershed. While the MADM model proposed here is relatively complex, implementation can be facilitated by embedding the model in a decision support system such as WAMADSS (Fulcher, 1996; Zhou et al., 1996). A decision support system automatically calculates the attributes of LWRMS and prompts property managers for information needed to identify preferences for attributes and the most preferred LWRMS for all properties in the watershed. In addition, a watershed alliance can use a decision support system to evaluate the sustainability of LWRMS selected by property managers and to identify economic and financial incentives that are most likely to enhance sustainable resource management in the watershed.

The selection of LWRMS by a property manager involves five steps as illustrated in Fig. 1. Numbers in each box refer to the steps that follow. First, the technically feasible LWRMS are identified for a property based on economic and biophysical considerations. For example, a private land manager is not likely to use a LWRMS that includes the planting of a crop that is not agronomically suited to local conditions even if the crop significantly reduces soil erosion and sediment delivery to nearby streams. Second, the property manager identifies the attributes for evaluating LWRMS. Third, values of the attributes for all technically feasible LWRMS are simulated using economic and biophysical models. Fourth, socially acceptable ranges of attributes are specified based on the property manager's preferences for attributes which are influenced by cultural, legal–institutional, ecological and other factors. For example, a manager of a national park is not at liberty to reduce surplus wildlife through private hunting because private hunting is illegal in national parks. Fifth, the efficient combinations of attributes for the property are derived. Sixth, the property manager selects the most preferred combination of attributes and LWRMS from the efficient ones.

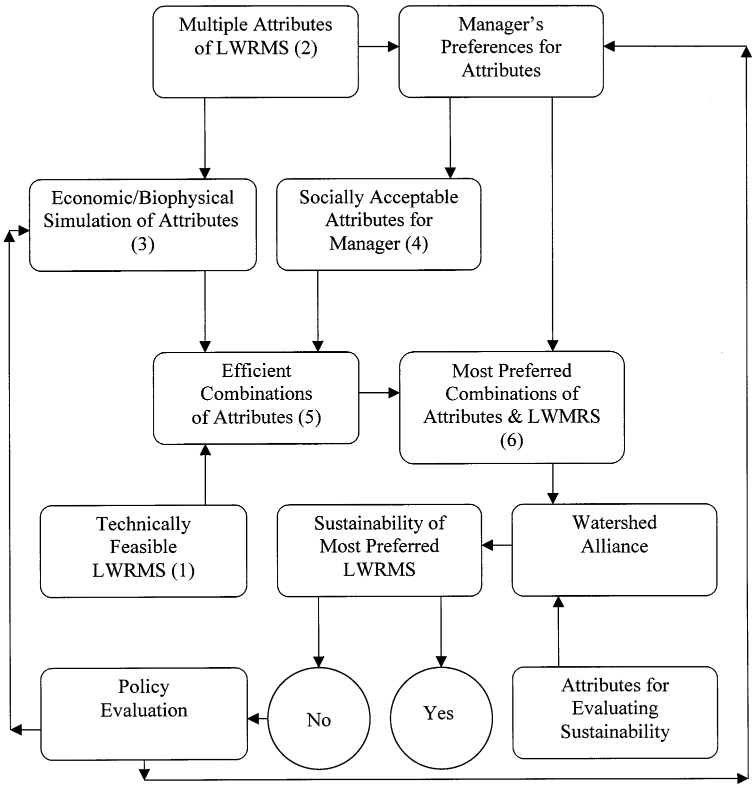

Fig. 1. Schematic of MADM approach to property management and watershed sustainability assessment. Numbers refer to six steps in MADM procedure (see paragraph 2 of Section 6).

6.1. Efficient combinations of attributes

This section describes the derivation of the efficient combinations of attributes and associated LWRMS. Suppose the property manager chooses n attributes to evaluate m technically feasible LWRMS for a property. Let $\mathbf{z}$ denote a column vector of the expected amounts of the n attributes provided by the m LWRMS, $\mathbf{A}$ denote a matrix the ith row of which indicates the amounts of the ith attribute per acre of the m LWRMS, $\mathbf{x}$ denote a column vector of the property acreage devoted to each of the m LWRMS and $\mathbf{u}$ denote a column vector of stochastic elements of $\mathbf{z}$. In terms of LWRMS for crop production, stochastic elements include weather patterns, occurrence of pest and disease infestation and variability in overall growing conditions. The relationship between $\mathbf{z}$ and $\mathbf{x}$ is:

$$\mathbf{z} = \mathbf{A}\mathbf{x} + \mathbf{u}$$

Since the acreage devoted to LWRMS cannot exceed the total acreage for the property, $\mathbf{x}$ must satisfy the constraint $\kappa'\mathbf{x} = X_{\mathrm{T}}$ where κ is a unit vector and X_{T} is total property acreage. Elements of $\mathbf{u}$ are assumed to be normally independently distributed (NID) with mean 0 and variance σ_i^2. Therefore, z_i (ith element of $\mathbf{z}$) is NID with mean $\mathbf{a}_i'\mathbf{x}$ and variance σ_i^2 where $\mathbf{a}_i'$ is the ith row of $\mathbf{A}$. The elements of $\mathbf{A}$ and σ_i^2 are estimated from the simulated values of the attributes (see Section 8).

If attributes are not normally distributed, it may be possible to achieve normality by transforming the original data (Emerson and Stoto, 1982). For example, Scott et al. (1992) used the square root function ($z^{1/2}$) to transform water quality data that was not normally distributed.

Efficient combinations of attributes are combinations provided by feasible LWRMS that fall in the socially acceptable ranges specified by the property manager. The latter are defined as follows:

$$\Pr\{z_i \geq z_i^*\} \geq 1 - \alpha_i \ (i = 1, \ldots, n) \tag{1}$$

where z_i^* is the minimum acceptable value of z_i, $0 \leq \alpha_i \leq 1$ and $(1 - \alpha_i)$ is the reliability with which $z_i \geq z_i^*$ is achieved. z_i^* and α_i are chosen by the property manager.

Since Eq. (1) cannot be directly used as a constraint in an optimization model, it is transformed to its statistical equivalent based on the properties of z_i. Specifically, when z_i is NID($\mathbf{a}_i'\mathbf{x}$, σ_i^2), Eq. (1) can be written as:

$$\Pr\{[(z_i - \mathbf{a}_i'\mathbf{x})/\sigma_i)] \geq [(z_i^* - \mathbf{a}_i'\mathbf{x})/\sigma_i)]\} \geq 1 - \alpha_i \ (i = 1, \ldots, n) \tag{2}$$

or equivalently,

$$F(y_i) = \alpha_i,$$

where $F(y_i)$ is the cumulative standard normal distribution function and $y_i = (z_i^* - \mathbf{a}_i'\mathbf{x})/\sigma_i$.

Solving for z_i^* in the expression for y_i gives:

$$z_i^* = \mathbf{a}_i'\mathbf{x} + y_i\sigma_i$$

Substituting this expression for z_i^* into Eq. (2) and rearranging terms gives:

$$\Pr(z_i \geq \mathbf{a}_i'\mathbf{x} + y_i\sigma_i) \geq 1 - \alpha_i \ (i = 1, \ldots, n) \tag{3}$$

Eq. (3) can be rewritten as:

$$z_i \geq \mathbf{a}_i'\mathbf{x} + y_i\sigma_i \ (i = 1, \ldots, n) \tag{4}$$

where y_i is chosen such that $F(y_i) = \alpha_i$.

The efficient combinations of attributes are determined by using Eq. (4) to formulate a stochastic version of the ε-constraint method (Haimes et al., 1971; Haimes and Hall, 1974; Cohon, 1978; Cohon and Marks, 1993). This method requires maximizing a primary attribute subject to chance constraints on the secondary attributes. Designation of primary and secondary attributes does not alter the solution. For example, when $n = 3$, z_1 is maximized subject to chance constraints on z_2 and z_3, namely:

$$\max z_1 = \mathbf{a}_1'\mathbf{x} + y_1\sigma_1 \text{ subject to: } z_i \geq \mathbf{a}_i'\mathbf{x} + y_i\sigma_i \ (i = 2, 3) \text{ and } \kappa'\mathbf{x} = X_T \tag{5}$$

The optimization problem given in Eq. (5) is similar to the chance-constrained linear programming problem formulated by Prato and Wu (1995) to derive efficient LWRMS for an agricultural watershed in north central Missouri. Using the approach of Prato and Wu (1995), Eq. (5) would be solved by fixing y_1 and solving the problem for alternative values of y_2 and y_3. Specifically, the values of $\mathbf{x}$ for the m LWRMS are substituted into $y_i = (z_i^* - \mathbf{a}_i'\mathbf{x})/\sigma_i$ to obtain a range of values for y_1, y_2 and y_3. Each range of values is divided into intervals. The value of y_1 is fixed at the midpoint of one of the intervals for y_1 and Eq. (5) is solved for all combinations of the midpoints of the intervals for y_2 and y_3. The procedure is repeated for all midpoint values of y_1.

Since $\mathbf{z}$ is stochastic, different combinations of LWRMS can result in the same efficient combination of attributes. Hence, there is not a one-to-one correspondence between $\mathbf{z}$ and $\mathbf{x}$ as there is in non-stochastic applications of the ε-constraint method such as those made by Ma (1993) and Xu et al. (1995). Suppose there is an efficient set of LWRMS corresponding to each efficient combination of attributes. If the property manager is indifferent toward LWRMS in the efficient set, then any LWRMS selected at random from that set will be equally preferred by the manager. However, if the property manager has preferences for LWRMS in the efficient set, then it may be possible for the manager to select a most preferred LWRMS corresponding to each efficient combination of attributes.

A hypothetical illustration of the efficient combinations of two attributes is given in Fig. 2. For simplicity, it is assumed that $\alpha_1 = \alpha_2 = 0$ which implies that z_1 and z_2 must always exceed z_1^* and z_2^*, respectively. Efficient combinations of z_1 and z_2 fall in the area ABCD. Combinations of z_1 and z_2 in the area BCE (excluding combinations on BC) are not efficient because combinations in the area ABCD dominate them. Specifically, every combination in area ABCD has more of z_1 and/or z_2 than every combination in the area BCE.

6.2. Preferred combinations of attributes

There are several ways to derive the most preferred combination of attributes from the efficient combinations of attributes. In the expected utility method, the most preferred combination of attributes is determined by maximizing an additive utility function in the attributes (Prato, 1997). In many applications, the utility function is specified as a weighted sum of attributes. The expected utility method requires specifying the utility function and attribute weights. In cases where the property manager only specifies the relative importance (rank order) of attributes, it is possible to rank the efficient LWRMS (Yakowitz et al., 1993).

In the surrogate worth tradeoff method, information provided by the property manager is used to derive the most preferred combinations of attributes (Haimes and Hall, 1974, 1977). The free iterative search procedure requires the property manager to search the efficient combinations of attributes in an iterative manner to find the most preferred one (Tecle et al., 1994). The analytical hierarchy process derives weights for attributes from paired comparison importance ratings between attributes made by the property manager (Saaty, 1987).

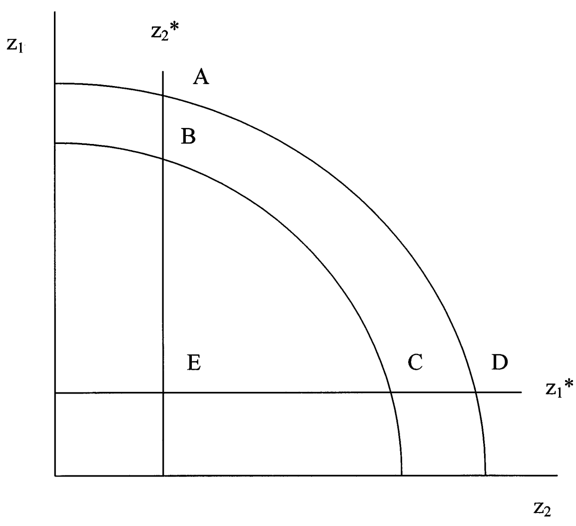

Fig. 2. Efficient combinations of attributes z_1 and z_2 (area ABCD).

In the Aspiration–Reservation Based Decision Support System (ARBDSS), the efficient combinations of attributes for a property and the management goals of the property manager are combined in a mathematical programming problem that is solved for the most preferred combination of attributes and associated LWMRS (Makowski, 1994; Fischer et al., 1996). The ARBDSS requires the property manager to express the management goals for a property in terms of the aspiration (most preferred) and reservation (least preferred) values for each and every attribute and how each attribute is to be treated (maximized, minimized or targeted to some value).

Stochastic dominance with respect to a function (SDWF) is a statistical method for classifying LWRMS as non-dominated (efficient) or dominated (inefficient) based on the cumulative probability densities for attributes and the landowner's risk attitudes (Whitmore and Findlay, 1978). Risk attitudes are characterized by the upper and lower bounds on the risk aversion function (Williams, 1988). This function defines both the nature (risk averse, neutral or prone) and degree (high or moderate) of risk for a decision-maker (Keeney and Raiffa, 1976). Application of the SDWF method is more general and entails fewer assumptions than other methods.

Attribute weights are subjective in nature because they are determined from information provided by the property manager. Some analysts consider subjective weights to be inferior to statistically derived weights because decision-makers lack, "sufficient insight into the parameters of their quasi-rational… [evaluations]… to generate the weights they applied" (Cooksey, 1996, p. 171). In judgment analysis, attribute weights are estimated using a multiple regression equation in which the scores a property manager assigns to different LWRMS are regressed on the corresponding attributes of those LWRMS. The relative importance of attributes is evaluated based on several statistical measures including unadjusted regression coefficients, beta coefficients, usefulness coefficients, hierarchical contribution coefficients and others.

7. Sustainability of LWRMS

While the property is an appropriate scale for modeling the selection of LWRMS by property managers, the watershed is a suitable scale for assessing the sustainability of LWRMS. Suppose a watershed alliance composed of stakeholders selects R attributes for evaluating the watershed-scale sustainability of the most preferred LWRMS selected by property managers (Fig. 1). Attributes chosen by the alliance are expected to differ from those selected by property managers. For example, while net income is a relevant and important economic attribute for a property manager, regional income and employment are more suitable attributes for evaluating economic sustainability at the watershed scale. Similarly, biodiversity is likely to be more important to the coalition than to property managers.

Let $\mathbf{z}^{\mathrm{A}}$ be a vector of the R attributes provided by the most preferred LWRMS for a watershed. Since $\mathbf{z}^{\mathrm{A}}$ is stochastic, sustainability is evaluated in terms of the relative frequency of achieving the minimum acceptable values of the R attributes, namely $\mathbf{z}^{\mathrm{S}}$, established by the watershed alliance. Two forms of sustainability can be evaluated, strong and weak (Pearce et al., 1990). Strong sustainability requires the relative frequency with which each and every attribute exceeds a minimum acceptable value to be greater than or equal to a reliability level established by the alliance, namely:

$$\mathrm{f}(\mathrm{z}_r^{\mathrm{A}} \geq \mathrm{z}_r^{\mathrm{S}}) \geq \beta_r \text{ where } r = 1,\ldots, R \tag{6}$$

z_r is the rth attribute of $\mathbf{z}^{\mathrm{A}}$, $\mathrm{z}_r^{\mathrm{S}}$ is the rth attribute of $\mathbf{z}^{\mathrm{S}}$, $\mathrm{f}(\mathrm{z}_r^{\mathrm{A}} \geq \mathrm{z}_r^{\mathrm{S}})$ is the relative frequency with which $\mathrm{z}_r^{\mathrm{A}}$ exceeds $\mathrm{z}_r^{\mathrm{S}}$ and β_r is a reliability level $(0 \leq \beta_r \leq 1)$.

Weak sustainability requires the relative frequency with which the expected value of a composite index of attributes exceeds a minimum acceptable value to be greater than or equal to a reliability level established by the alliance, namely:

$$\mathrm{f}[E(CI^{\mathrm{A}}) \geq CI^{\mathrm{S}}] \geq \theta \tag{7}$$

where $E(CI^{\mathrm{A}})$ is the expected value of the index of the attributes contained in $\mathbf{z}^{\mathrm{A}}$, CI^{S} is the value of the composite index when each attribute is set equal to its minimum acceptable value and θ is a reliability level $(0 \leq \theta \leq 1)$. The weak sustainability criterion is compensatory in the sense that high values of one attribute can compensate for low values of other attributes. Prato (1998a) gives a more complete discussion of weakly and strongly sustainable resource management. The relative frequencies given in Eqs. (6) and (7) are estimated using simulated values of the attributes (see Section 8).

Members of the alliance might pick different attributes, minimum acceptable values of attributes and reliability levels for assessing sustainability. This could lead to disagreements among alliance members about whether or not the most preferred LWRMS are sustainable. Such disagreements can be resolved using conflict resolution procedures such as the ones developed by Hafkamp and Nijkamp (1993) and Bentham and Greer (1998).

8. Evaluation of attributes

Implementation of the MADM model and sustainability assessment requires evaluation of the attributes. In particular, the simulated values of the attributes are needed to estimate the means and variances of the attributes and the relative frequencies in Eqs. (6) and (7). There are several models that are suitable for evaluating the social, economic, environmental and ecological attributes of agricultural LWRMS. Social implications of LWRMS can be evaluated using social impact analysis. Economic impacts can be evaluated using the Cost and Return Estimator (CARE) (USDA, 1988). CARE estimates the gross income, cost and net income for LWRMS. Regional income and employment effects of changes in property income can be estimated using the IMpact analysis for PLANning (IMPLAN) model. IMPLAN is an input–output computer model developed by the US Forest Service to evaluate changes in total economic output, total household income and employment for expenditure changes in up to 528 sectors of an economy.

CREAMS (Knisel, 1980) and EPIC (Williams et al., 1990) simulate the environmental effects of

LWRMS at the field scale. ANSWERS (Beasley and Huggins, 1982), AGNPS (Young et al., 1989) and SWAT (Arnold et al., 1993) simulate biophysical effects of agricultural management systems in a watershed. WAMADSS is a watershed decision support system that allows interactive evaluation of the economic and environmental impacts of LWRMS at the watershed scale (Fulcher, 1996; Zhou et al., 1996). WAMADSS uses CARE to estimate costs and returns for LWRMS at the property scale and AGNPS or SWAT to simulate biophysical effects at the watershed scale.

Evaluating the watershed–stream biota responses to changes in LWRMS is complex, but essential for determining whether natural resources are being managed in a sustainable manner at the watershed scale. Effects of LWRMS on aquatic ecosystems have received more attention than effects on terrestrial ecosystems, particularly in agricultural watersheds. LWRMS in upland areas of a watershed affect environmental conditions (water quality, sedimentation and temperature) in the stream channel which, in turn, affect stream biota (Ryder and Karr, 1989; Rabeni, 1992). Relatively little research has been done relating upland LWRMS to stream biota. The few studies that address this issue deal primarily with forested watersheds (Joyce et al., 1990).

Stream–habitat models are available for simulating how stream-dwelling organisms respond to changes in key environmental variables such as siltation and chemical pollution (Fausch et al., 1988). Quantitative relations between fish/invertebrate communities and siltation levels (Berkman and Rabeni, 1987; Rabeni and Smale, 1995) and between fish communities and dissolved oxygen and summer temperatures (Smale and Rabeni, 1991) have been determined in selected locations. Ecological effects of proximate habitat conditions in a stream can be evaluated using an index of biological integrity that assesses the biological impacts of one or more environmental variables (Karr et al., 1986). Many of the outputs from watershed-scale biophysical simulation models can be used as inputs to stream–habitat assessment models.

In larger ecosystems, the CENTURY model has been used to evaluate how changes in climate and management could affect net primary production, nutrient availability and carbon/nitrogen fluxes (Parton et al., 1995). Based on the CENTURY model, Oijima and Parton (1996) found that that during the past 100 years, crop yields in the Great Plains increased whereas soil organic matter content and mineralization rates for soil nitrogen decreased, and Parton et al. (1995) and Donigian et al. (1995) concluded that alternative LWRMS in the Corn Belt would improve carbon storage in soils while causing only minor losses in corn yield.

9. Policy evaluation

If the watershed alliance determines that the most preferred LWRMS for properties in the watershed are not sustainable, then it is appropriate to evaluate alternative policies for stimulating the adoption of sustainable LWRMS. Policy evaluation involves three steps. First, the variables and relationships influenced by the policy are modified. Second, the new set of most preferred LWRMS for properties in the watershed are determined by reapplying the MADM procedure. Third, the sustainability of the new LWRMS is evaluated.

To illustrate this approach, consider a watershed alliance that evaluates the strong sustainability of LWRMS based on three attributes, watershed income, soil conservation and biodiversity. Consider a case in which the strong sustainability criterion, $f(z_r^A \geq z_r^S) \geq \beta_r$, is satisfied for watershed income but is violated for soil conservation and biodiversity. In this case, the most preferred LWRMS for the watershed are not strongly sustainable.

Suppose the watershed alliance wants to evaluate two policies for enhancing soil conservation and biodiversity. The first policy provides free technical assistance to property managers who adopt LWRMS that achieve sustainable levels of soil conservation and biodiversity, referred to as sustainable LWRMS. The second policy offers technical assistance plus cost sharing to property managers who adopt sustainable LWRMS. Cost sharing is justified based on the user pays principle which states that all stakeholders who benefit from adoption of sustainable LWRMS should share in their cost (Murray-Darling Basin Commission, 1996). Since society benefits from sustainable resource management, it should be willing to pay part of the private cost of implementing sustainable LWRMS. While difficult to achieve in practice, cost-sharing payments received by property managers should be proportional to the social benefits achieved by those systems.

Before recommending a policy, the watershed alliance should evaluate the potential impacts of various policies. Suppose evaluation of the first policy (technical assistance) indicates that the policy causes property managers to select most preferred LWRMS that increase soil conservation and biodiversity, but not enough to achieve the minimum acceptable values for those attributes. The first policy is insufficient to achieve sustainable resource management. The second policy (technical assistance plus cost sharing) has a greater likelihood of achieving sustainable resource management than the first policy because it reduces the private cost of using sustainable LWRMS which increases the net income generated by those systems. Higher net income might stimulate some property managers to switch from non cost-shared to cost-shared LWRMS. Land and water resource management in the watershed would be sustainable if a sufficient number of property managers made this switch.

The cost effectiveness of the two policies can be evaluated using a composite index. Let the expected value of the composite index of attributes be $E(CI_1^{\mathrm{A}})$ for the first policy and $E(CI_2^{\mathrm{A}})$ for the second policy. One measure of cost effectiveness is:

$$[E(CI_i^{\mathrm{A}}) - E(CI_0^{\mathrm{A}})]/G_i \ (i = 1,\ 2)$$

where $E(CI_0^{\mathrm{A}})$ is the expected value of the composite index with no policy, $E(CI_i^{\mathrm{A}}) \geq E(CI_0^{\mathrm{A}})$ and G_i is the government cost of implementing the ith policy. This ratio measures the overall expected improvement in sustainable resource management per dollar of government expenditure on policy implementation.

10. Conclusions

Implementation of an ecosystem approach to natural resource management raises concerns about the usefulness of conventional economic approaches that assign values to ecological services (contingent valuation) and evaluate the efficiency of preserving and restoring those services (cost–benefit analysis). Limitations of contingent valuation include its single-attribute nature, ethical objections and methodological problems with assigning monetary values to ecological services, pitfalls of separating value estimation from resource planning and management, inherent bias toward decisions that favor economic as opposed to ecological considerations and others.

Cost–benefit analysis has limitations for evaluating ecosystem management. Its weaknesses include the discounting of benefits and costs which can result in under-investment in ecological protection and restoration, the general practice of ignoring intra-generational and inter-generational fairness, non-monetary ecological consequences and uncertainty in benefits and costs, incompatibility with community-based collaborative decision-making and others.

Multiple attribute decision making (MADM) is an alternative conceptual framework for evaluating and selecting LWRMS for individual properties that alleviates several of the limitations of contingent valuation and cost–benefit analysis. Advantages of MADM are that it facilitates public participation and collaborative decision-making, avoids some of the ethical, theoretical and practical shortcomings of cost–benefit analysis, does not require assigning monetary values to ecological services, allows consideration of multiple attributes, is not culturally biased and others.

The MADM model described in this paper explains how a property manager selects the most preferred LWRMS for a property based on their multiple stochastic attributes. Application of the model requires determination of the technically feasible LWRMS for a property and specification of the socially acceptable ranges of attributes. This information is combined with economic/biophysical simulations to derive the efficient combination of attributes and LWRMS for a property.

The property manager then selects the most preferred combination of attributes from the efficient combinations of attributes for a property using utility maximization, surrogate worth tradeoff, free iterative search, analytical hierarchy process, Aspiration–Reservation Based Decision Support System or stochastic dominance.

A watershed alliance can evaluate the sustainability of the most preferred LWRMS for properties in a watershed based on a weak or strong sustainability criterion. If the alliance determines that the most preferred LWRMS are not sustainable, then an index of attributes is used to evaluate the cost effectiveness of alternative public policies for stimulating the adoption of more sustainable LWRMS.

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