Journal of Environmental Management (2000) **60**, 325–337
doi:10.1006/jema.2000.0387, available online at http://www.idealibrary.com on IDEAL®

Multiple attribute evaluation of landscape management

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Economic approaches to the valuation of ecological services have several limitations. Some of these limitations can be overcome using the multiple attribute decision-making model developed in this paper. The model postulates that a private or public decision-maker selects a site/landscape management plan based on the biophysical and economic attributes of alternative management plans, the decision-maker's preferences for attributes, and constraints on the selection of a management plan.

Two cases are examined. Case A is a watershed consisting of publicly owned land that is managed at the site, management unit and landscape scales. Management is based on the philosophy of ecosystem management. Case B is a watershed composed of several privately owned units that are managed at the site scale by decision-makers whose primary motivation is economic profit. The preferred management plan in both cases is determined using a two-stage procedure. The first stage uses a stochastic programming model to identify the most efficient management plans for a site/landscape. The second stage determines which efficient management plan for a site/landscape is preferred by maximizing an expected utility function that is additive in the attributes and assumes that the decision-maker is risk neutral.

Whether a land-management plan results in strongly or weakly sustainable resource conditions is evaluated. Strong sustainability requires the probability of exceeding the minimum acceptable value of an attribute to be greater than or equal to a pre-determined reliability level for each attribute. Weak sustainability requires the same condition except that it applies to a composite index of the attributes rather than each attribute. Bayes theorem is used to evaluate uncertainty about whether the state of a landscape is sustainable.

Keywords: multiple attribute decision-making, site and watershed management, sustainable resource management.

Introduction

Most studies of ecosystem management and restoration address a particular biophysical, socio-economic or cultural process related to landscape structure and function (Agee and Johnson, 1988; Kohm and Franklin, 1997; Moore *et al*., 1994; Naiman *et al*., 1992; National Park Service, 1994; Williams *et al*., 1997). Integrated biophysical, socioeconomic and ecological assessments are usually made using simulation models (Flamm and Turner, 1994; Fulcher, 1996; Negahban *et al*., 1993; Pastor and Johnson, 1992; Prato *et al*., 1996a, Tecle *et al*., 1994; Zhou, 1996). Economic valuation of ecological services, which is a primary component of integrated assessment, typically employs methods such as avoided costs, contingent valuation, cost-benefit analysis, input-output analysis, natural resource accounting and substitution costs (Costanza *et al*., 1998; Goulder and Kennedy, 1997; Prato, 1998a, Wilcox and Harte, 1997).

These methods have certain limitations. For example, cost-benefit analysis has been criticized because it: (1) discounts benefits and costs which can result in under-investment in ecological protection and restoration; (2) typically ignores intra-generational and inter-generational fairness; (3) excludes non-monetary ecological consequences and uncertainty in benefits and costs; and (4) is not particularly compatible with community-based decision-making. Non-market valuation methods, and contingent valuation (CV) in particular, have been criticized because: (1) a single-attribute valuation technique is poorly suited for evaluating the multifaceted ecological

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Received 1 October 1998; accepted 15 September 2000

0301–4797/00/120325+13 $35.00/0

impacts of resource management decisions; (2) assigning monetary values to ecological services is rejected by many non-economists on ethical and moral grounds; (3) willingness-to-pay (WTP) measures used in CV studies are likely to be biased by imperfect information on the part of the respondent, embedding of the value of other goods in stated WTP values and other response biases; and (4) survey respondents in CV studies 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 (Kahn, 1996; Sagoff, 1988). These and other limitations of economic valuation methods provide impetus for seeking new ways to evaluate the benefits of ecological services.[1]

The primary purpose of this paper is to examine the usefulness of multiple attribute decision-making (MADM) in selecting management plans at the site or landscape scales and determining the sustainability of those plans at the landscape scale. A MADM approach postulates that a decision-maker selects a management plan for an area based on its biophysical features, financial and economic conditions, preferences for attributes of plans, and other relevant constraints. MADM methods have been proposed and/or used to evaluate the management of water resources systems (Haimes and Hall, 1974), environmental resources (Janssen, 1992), food security (Haettenschwiler, 1994), forests (Penttinen, 1994), environmental impacts of agricultural production (Xu *et al*., 1995), regional water quality (Makowski *et al*., 1995), agroecosystems (Prato *et al*., 1996a), wildlife (Prato *et al*., 1996b), soil and water resources (Prato, 1998c) and other resources (El-Swaify and Yakowitz, 1998). This paper makes three contributions: (1) it extends the application of MADM to the selection of management plans at the property or watershed scales; (2) it demonstrates the usefulness of MADM in assessing the weak and strong sustainability of management plans at the watershed scale; and (3) it illustrates how Bayes theorem is used to deal with uncertainty regarding whether implementation of a management plan results in sustainable socioeconomic and ecological conditions.

Spatial and temporal scales

Ecosystem management requires an understanding of how socioeconomic and ecological processes operate at relevant spatial and temporal scales. Three spatial scales are common in ecosystem management, namely site (4–200 ha), landscape (200–4000 ha) and region (thousands of square kilometers) (Schleusner, 1994). A site is a geographic area having relatively homogenous landforms, soil types and climate. Agricultural fields and timber stands are sites. A mosaic or patchwork of sites is a landscape. Watersheds are also landscapes. Landscapes that have similar soils, landforms and physiographic features constitute a region (Bailey, 1988). Superimposed on these three spatial scales are management units that range in size from multiple sites to multiple landscapes. A farm, ranch or forest is a management unit.

Ecosystem management of sites requires knowledge of how ecological processes and structures interact on those sites. Ecological and other services provided by landscapes are determined by the spatial and temporal arrangement of site management plans on the landscape (Allen, 1994). While most management efforts focus on the unit and landscape scales, broad-based ecosystem assessment occurs at the regional scale. For example, the forest management plan recommended by the Forest Ecosystem Management Assessment Team was regional in scope (FEMAT, 1993).

A private or public decision-maker manages the sites located within a management unit. A farmer manages the fields on a farm, a rancher manages the grazing paddocks on a ranch and a forester manages the stands in a forest. The same management unit might include all three land uses (farming, grazing and forestry). Several decision-makers commonly manage landscapes dominated by private land. In

[1] Ecosystem management is a 'method for sustaining and restoring natural systems and their functions and values' (Interagency Ecosystem Management Task Force, 1995:3). Ecosystem services are 'the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life' (Daily, 1997:3).

contrast, a single decision-maker, such as a public agency, typically manages landscapes dominated by public land. This is significant because it is more difficult to achieve sustainable resource management when there are numerous decision-makers than when there is only one decision-maker.

In the spatial dimension, rural landscapes typically include several farms, ranches and forests managed by different decision-makers. However, a large farm, ranch or timber operation can cover several landscapes. In addition, federal and state agencies often manage forestlands and natural areas that cover more than one landscape. Regions cover multi-state areas that include units managed by many private and public decision-makers. The complexity of resource management, number/diversity of decision-makers and resource conflicts generally increase with spatial scale (site to landscape to region). For example, conflicts regarding methods of restoring salmon populations in the Columbia River Basin of North America increase when moving to larger scales, such as individual watersheds to major tributary rivers to the entire basin.

The temporal dimension of landscape management is also important. Sustainable management of soil and water resources requires that soil and water conservation be sufficient to maintain long-term resource productivity. When soil erodes at rates that exceed regeneration and water is contaminated beyond its capacity to assimilate pollutants, food production, human health and biodiversity are adversely affected.

Spatial and temporal dimensions of landscape management interact. Parton *et al*. (1995) and Donigian *et al*. (1995) found that adoption of alternative farming systems in the US Corn Belt would improve carbon storage in soils while causing only minor losses in corn yield. Nitrogen losses in runoff from agricultural areas in the Midwestern US have increased hypoxia (low oxygen conditions) in the Gulf of Mexico. Deforestation in the Amazon Basin has reduced carbon sequestration and contributed to global warming (Barbier, 1994). These examples demonstrate the important linkages between the spatial and temporal dimensions of landscape management.

MADM Framework

This section describes a MADM-based conceptual model of landscape management. Consider a watershed (landscape) composed of K sites each having relatively homogenous social, cultural and environmental characteristics (Figure 1). Sites are delineated based on land cover/use, soil type, hypsography, hydrography and other spatial characteristics codified using geographic information systems and remote sensing techniques. Management units are delineated using land ownership information.

Land ownership is a significant distinguishing feature of landscape management. For this reason, two ownership-based cases are evaluated. In case A, land in the watershed is publicly owned. One decision-maker, such as a public agency, is responsible for management decisions at the site, management unit and landscape scales. Case A is typical of protected and natural areas, such as a national park or national forest. While ecosystem management is the underlying paradigm for case A, other management paradigms can be evaluated by incorporating plan attributes relevant to those paradigms. In case B, the watershed is dominated by privately owned properties that are managed at the site and management unit scales by decision-makers whose primary motivation is economic profit. Case B fits the conditions in predominately agricultural, forested and prairie landscapes, which are

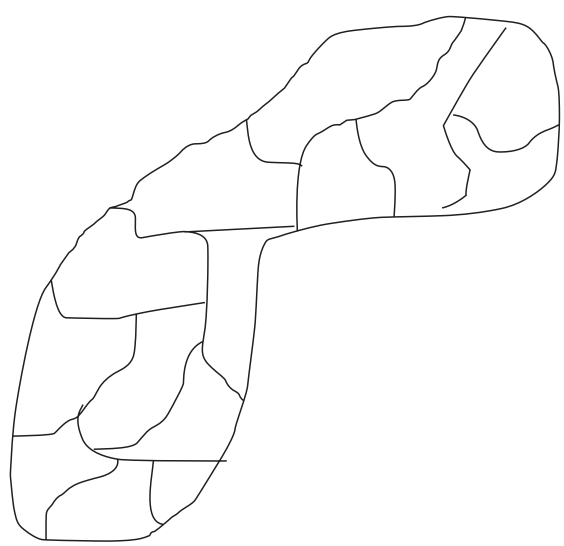

Figure 1. Landscape composed of K sites.

common throughout the world. Large landscapes typically combine elements of cases A and B resulting in a mix of public and private lands. This is important because some of the most challenging land-use conflicts occur in areas adjacent to the boundary between private and public lands. Landscapes vary throughout the world from the heavily populated urban landscapes of Western Europe, India and China, to the sparsely populated rural landscapes of central and southern Africa, northern China and the western US.

Cases A and B involve selection of a site management plan (SMP). A SMP refers to the land-use/management practices applied at a site as well as the planning horizon over which the practices are applied. A SMP for a private timber stand specifies timber production and harvesting methods including mix of tree species, pest control, type and extent of road and bridge construction, harvest methods and length of rotation. A SMP for an agricultural field designates the crop rotation, tillage method and nutrient-chemical application rates. In a given planning horizon, a decision-maker is allowed to select one SMP per site. However, site management can be altered in the next planning horizon. For example, an agricultural site can be converted to a timber site, or *vice versa*.

When resource management is motivated by ecological considerations, as in case A, selection of an SMP requires an understanding of ecological processes (microclimate, water and mass movement, soil development, nutrient cycles, food webs, disturbance regimes and others), ecological structures (biotic composition, vegetative patterns, stream morphology, distribution of course woody debris in stream channels and others) as well as the interactions between process and structure (Allen, 1994). Selection of a SMP is motivated by economic considerations in case B. However, soil erosion and water pollution impacts of a SMP are likely to be important to a private decision-maker in areas where resource degradation from these sources is significant.

The landscape-scale decision in case A is selection of a landscape management plan (LMP). A LMP is a particular spatial arrangement of SMP in a watershed. Ecological patterns and processes are managed at the landscape scale by varying the spatial arrangement of SMP (Franklin and Forman, 1987; Rykiel *et al.*, 1988). There is not a landscape management decision in case B because management decisions are made only for sites and management units. There is social concern about whether the LMP selected in case A and the LMP that results from the SMPs selected in case B are sustainable.

Cases A and B are likely to involve different plan attributes. Attributes of interest to the manager of a public forest (case A) include timber cut, income, employment, tax revenues, recreational values, fish and wildlife habitat, water quality, biodiversity and others. Attributes of importance to the manager of private agricultural/forest land (case B) include harvest rate, income and costs. In certain circumstances, environmental attributes of a management plan (soil erosion, water contamination and biodiversity) are important to private decision-makers. Conservation of biodiversity is an important attribute in landscapes containing threatened and endangered species. While in theory there is no limit to the number of attributes, a decision-maker's ability to evaluate attributes is likely to decrease with the number of attributes. Some analysts recommend against using more than seven attributes.

Attributes are competitive or complementary. For example, increasing timber harvest beyond some threshold rate is likely to decrease recreational opportunities, fish and wildlife habitat and biodiversity. Beyond the threshold, timber harvest is competitive with these other attributes. Conserving biodiversity is expected to increase recreational opportunities and improve fish and wildlife habitat. Hence, biodiversity is complementary with recreational values and fish and wildlife habitat.

The planning horizon for selecting a SMP or LMP can influence the relationship between attributes. For example, reducing timber harvest in an effort to increase biodiversity reduces short-term income. Yet, high biodiversity increases long-term income relative to its level with low biodiversity. Therefore, income and biodiversity are competitive in the short-term but complementary in the long-term.

In summary, there are fundamental differences in the nature of decision-makers, management units and attributes for cases A and B. There is a single public decision-maker

Table 1. Description of MADM applications

Case	Decision-maker	Decision variable	Management unit	Relevant attributes	Management objective
A	Public manager	SMP and LMP[a]	National forest, national park, wildlife refuge, etc.	Timber cut, income, employment, tax revenue, recreational value, water quality and biodiversity	Select preferred SMP and LMP
B	Private managers	SMP	Farm, ranch or forest	Harvest rate, income and costs	Select preferred SMP

[a]SMP is a site-management plan and LMP is a landscape-management plan.

in case A and several private decision-makers in case B. Selection of a SMP is relevant in both cases. A LMP is selected in case A, but not in case B. Sustainability of a LMP is important in both cases. Decision-makers are assumed to have the same general objective, namely to select a management plan that provides the most efficient and preferred combination of attributes subject to relevant biophysical, financial and economic constraints.

MADM is a suitable framework for selecting a SMP and LMP and for evaluating the sustainability of a LMP. In brief, this paper considers only a subset of MADM applications that are possible in cases A and B. In particular, MADM is discussed in terms of selecting the preferred LMP for case A and in evaluating the sustainability of the LMP that results in case B. Focusing on landscape applications of MADM is consistent with the theme of this paper. The MADM applications discussed here are summarized in Table 1.

Case A

Based on previous work (Prato, 1999; Prato and Hajkowicz, 1999), selection of the preferred LMP for a watershed is modeled using a two-stage procedure. The first stage identifies the most efficient combinations of attributes and associated LMP for the watershed and the second stage determines the preferred LMP from among the efficient LMP identified in the first stage.

First stage

Efficient combinations of attributes are determined by solving an optimization problem that is a stochastic version of the ε-constraint method (Cohon, 1978; Cohon and Marks, 1993; Haimes *et al.*, 1971; Haimes and Hall, 1974). In this method, a primary attribute is maximized subject to chance constraints on secondary attributes. Designation of primary and secondary attributes does not alter the solution.

Suppose a public planning activity identifies R feasible LMP for a landscape. LMP are evaluated based on I attributes. The relationship between the I attributes and R LMP is:

$$\begin{bmatrix} z_1 \\ \cdot \\ \cdot \\ z_I \end{bmatrix} = \begin{bmatrix} a_{11} \dots\dots a_{1R} \\ \dots\dots\dots \\ a_{I1} \dots\dots a_{IR} \end{bmatrix} \begin{bmatrix} LMP_1 \\ \cdot \\ + \\ LMP_R \end{bmatrix} + \begin{bmatrix} u_1 \\ \cdot \\ \cdot \\ u_I \end{bmatrix} \tag{1}$$

or equivalently:

$$z = Ax + u \tag{2}$$

where z is a column vector of the amounts of the I attributes, A is a matrix each row of which indicates the amount of a particular attribute provided by the R LMP, x is a column vector of the R LMP and u is a column vector of stochastic elements of z that include the effects of weather and natural disturbances (fire, pests and disease).

Each element of u is assumed to be normally independently distributed (NID) with mean 0 and variance σ_i^2. Therefore, $E(z)=Ax$ and the ith element of z, namely z_i, is NID with mean $a_i'x$ and variance σ_i^2 where a_i' is the ith row of A. a_i' and σ_i^2 are estimated using simulated or expert-determined values of the attributes. For example, attributes of agricultural management plans can be simulated using the Soil and Water Assessment Tool (Arnold *et al.*, 1993). In a given planning horizon, only one LMP can be selected for a watershed. Therefore, x must satisfy the constraints $\kappa' x=1$ where κ' is a unit vector, namely $\kappa'=(1, \ldots, 1)$, and x_i (ith element of x) equals 0 or 1 ($i=1, \ldots, I$).

When u is not normally distributed, it may be possible to transform the original data to be normally distributed (Emerson and Stoto, 1982). Scott *et al.* (1992) used the square root function ($z^{1/2}$) to transform water quality data that were not normally distributed.

The first-stage optimization problem is illustrated for three attributes of a LMP: watershed net return (M), soil erosion (S) and biodiversity (B); $z_1=M$, $z_2=S$ and $z_3=B$. If M is the primary attribute and E and B are the secondary attributes, then the problem is:

$$\begin{aligned} &\max M = a'_M x + y_M \sigma_M \\ &\text{subject to: } S \le a'_S x + y_S \sigma_S \\ &\qquad B \ge a'_B x + y_B \sigma_B \\ &\qquad \kappa' x = 1 \\ &\qquad x_i = 0, 1. \end{aligned} \tag{3}$$

$y_i = (z_i^* - a_i' x)/\sigma_i$ where z_i^* is the maximum acceptable value (in the case of soil erosion) or minimum acceptable value (in the case of biodiversity) of an attribute ($i=M$, S, B). y_i is defined so that $F(y_i)=\alpha_i$ where $F(y_i)$ is the cumulative standard normal distribution function for y_i, $0 \le \alpha_i \le 1$ and $(1-\alpha_I)$ is the reliability with which the value of an attribute exceeds its minimum acceptable value or falls below its maximum acceptable value. Lower values of α_i imply higher reliability. z_i^* and α_i are chosen by the decision-maker.

Attributes are standardized in order to avoid non-optimal solutions resulting from differences in the units of measurement for attributes. The standardized value of a positive attribute, such as net return and biodiversity, is $z_{di} = [z_i - \min(z_i)]/[\max(z_i) - \min(z_i)]$ where z_i is the raw value, $\min(z_i)$ is the minimum raw value and $\max(z_i)$ is the maximum raw value of the ith attribute (Tecle *et al.*, 1995). Maximum and minimum values are determined from the simulated or expert-determined values of z_i. A negative attribute such as soil erosion is standardized into a positive attribute such as soil conservation by $z_{di} = 1 - [z_i - \min(z_i)]/[\max(z_i) - \min(z_i)]$. A standardized attribute falls in the interval [0,1].

Replacing the first constraint in Equation (3) with $C \ge a'_C x + y_C \sigma_C$ (C is soil conservation), standardizing all attributes and solving gives the efficient combinations of M, C and B. Since z is stochastic, it is possible for more than one LMP to result in the same efficient combination of attributes. If this occurs, then there is not a one-to-one correspondence between z and x as in non-stochastic applications of the ε-constraint method (Ma, 1993; Xu *et al.*, 1995).

Suppose the solution to Equation (3) indicates there is an efficient set of LMP corresponding to each efficient combination of attributes. If the decision-maker is indifferent toward the LMP in an efficient set, then s/he is equally satisfied by any LMP randomly selected from that set. However, if the decision-maker has preferences for LMP in the efficient set, then s/he selects the preferred LMP from that set. Based on this procedure, it is possible to associate a preferred LMP with each efficient combination of attributes. The set of efficient combinations of attributes is designated Z_e and the associated set of preferred LMP is designated X_e.

Efficient combinations of attributes lie on the efficiency frontier such as the one illustrated in Figure 2. This hypothetical frontier is three-dimensional because there are three attributes (M, C and B). $C_d{}^*$ and $B_d{}^*$ are the standardized minimum acceptable values of C and B, respectively. To simplify the illustration of the efficiency frontier, it is assumed that $\alpha_C = \alpha_B = 1$ and that the attributes are competitive. In practice, it is more reasonable to expect C and B to be complements. Efficient combinations of attributes (Z_e) lie on the surfaces labeled PRS and PRK in Figure 2.

Second stage

In the second-stage, the decision-maker selects the preferred combination of attributes from the efficient combinations (Z_e) determined in the first stage. There are several ways to solve this problem. In the expected utility approach adopted here, the preferred combination of attributes is determined by maximizing an expected utility function subject to the constraint that the preferred combination is efficient, namely:

$$\text{maximize } E\{u[z_d(x)]\} \text{ subject to: } z_d \in Z_{de} \tag{4}$$

$u[z_d(x)]$ is the decision-maker's utility function, $z_d(x)$ are the standardized attributes, E

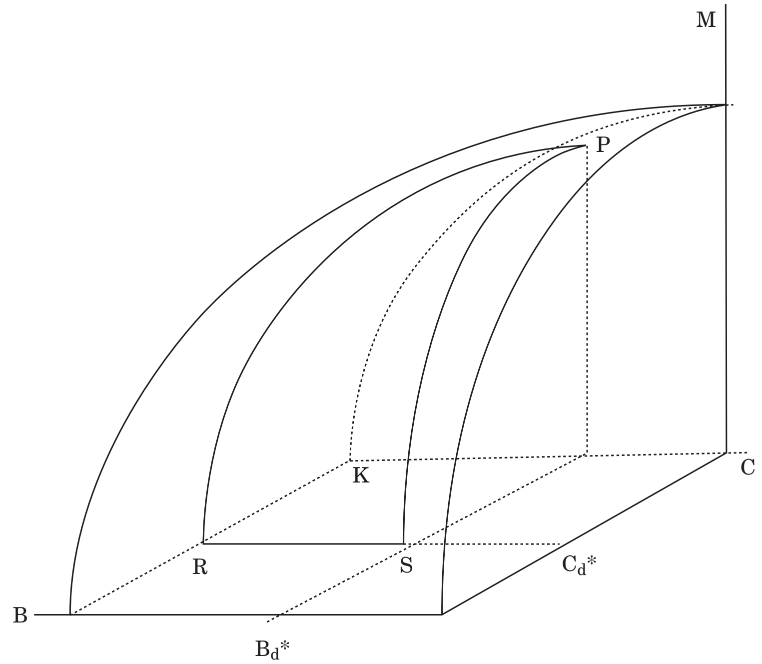

Figure 2. Hypothetical efficiency frontier for three attributes: net return (M), soil conservation (C) and biodiversity (B).

is the expected value operator and Z_{de} is the set of standardized efficient combinations of attributes determined in the first stage.

The most common specification of the expected utility function in MADM applications is the following weighted-additive form:

$$u[z_d(x)] = \sum_{i=M,\,C,\,B} k_i u_i[z_{di}(x)] \qquad (5)$$

where k_i is a positive scaling constant and the utility subfunction $u_i[z_{di}(x)]$ is an increasing function of z_{di}. Additive utility implies that attributes are mutually utility independent, or equivalently, that the marginal utility of one attribute is independent of the amounts of all other attributes ($\partial u_i(z_{di})/\partial z_{dj} = 0$ for all $i \neq j$). Additive multiattribute utility functions are very common in MADM applications (Yakowitz *et al.*, 1993; Foltz *et al.*, 1995; Tecle *et al.*, 1995) because of their simplicity and relevance to real world problems (Keeney and Raiffa, 1976).

When the utility function is additive, Equation (4) becomes:

$$\text{maximize } E\{u[z_d(x)]\} = \sum_{i=M,\,C,\,B} w_i E\{u_i[z_{di}(x)]\} \quad \text{subject to } z_d \in Z_{de} \qquad (6)$$

where w_i is the weight assigned to the ith attribute:

$$\sum_{i=M,\,C,\,B} w_i = 1 \text{ and } w_i \geq 0$$

Risk attitudes for the univariate utility subfunction, $u_i[z_{di}(x)]$, are characterized by the risk aversion function $r(z_{di}) = -u''(z_{di})/u'(z_{di})$, where $u'(z_{di})$ is the first derivative and $u''(z_{di})$ is the second derivative of the utility function with respect to z_{di} (Keeney and Raiffa, 1976:159–161). A decision-maker is risk neutral, risk averse or risk prone when r is zero, greater than zero or less than zero, respectively. Risk aversion is less restrictive than risk neutrality, but more difficult to apply in empirical studies. Risk prone preferences are not considered realistic except in gambling situations. For this reason, many MADM applications make the simplifying assumption that decision-makers have risk neutral preferences. When the utility function is additive in the attributes and the decision-maker is risk neutral, Equation (6) simplifies to:

$$\text{maximize } E\{u[z_d(x)]\} = \sum_{i=M,\,C,\,B} w_i[z_{di}(x)] \text{ subject to } z_d \in Z_{de} \qquad (7)$$

The preferred combination of attributes is found by solving Equation (7) for z_d. Associated with each efficient combination of attributes is an efficient LMP for the watershed. If attribute weights cannot be elicited from the decision-maker, then the efficient LMP identified in the first stage can still be ranked provided the decision-maker is able to rank the attributes (Yakowitz *et al*., 1993).

The above MADM model is admittedly complex and, in its present form, is not suitable for use by most decision-makers. Access to the model can be improved substantially by incorporating it in an interactive decision support system such as the Watershed Management Decision Support System (WAMADSS) (Fulcher, 1996; Zhou *et al*., 1996). The current version of WAMADSS utilizes a GIS-based biophysical-economic module to simulate economic and environmental attributes of alternative LMP. Adding a MADM module to WAMADSS would facilitate solution of the first and second-stage optimization problems. The MADM module would combine the simulated or expert-determined attributes with information supplied by the decision-maker regarding attribute weights (w_i), minimum and maximum acceptable values of attributes (z_i*) and reliability levels (α_i).

Case B

When a watershed contains only private property, the decision-makers are the property managers. In this case, the preferred SMP for each management unit (property) in the watershed is determined using the two-stage procedure described in the previous section. In the first stage, each property manager determines the most efficient combination of attributes and associated SMP for each site in a management unit. In the second stage, the decision-maker selects the SMP that provides the preferred combination of attributes for the site. The LMP consists of the preferred SMPs for all sites in the watershed and is designated LMP_W.

While LMP_W is optimal from a private viewpoint, it may not be optimal from a social viewpoint. Suppose a watershed alliance made up of watershed stakeholders wants to determine whether LMP_W is sustainable in terms of being compatible with ecosystem management. The latter has become popular in the US, where 18 federal agencies have adopted or are considering adoption of management programs based on an ecosystem approach (Haeuber and Franklin, 1996). For simplicity, the alliance is assumed to evaluate sustainability based on the same attributes given in case A, namely watershed net return (M), soil conservation (C) and biodiversity (B). In case B, property managers select SMP for management units in the watershed and the alliance decides whether or not the LMP_W implied by those SMP is sustainable.

Let z_P be a stochastic vector of the three attributes provided by LMP_W. Strong and weak sustainability of LMP_W are defined based on definitions given in the sustainable development literature (Pearce *et al*., 1990). Strong sustainability requires the probability of exceeding the minimum acceptable value of an attribute to be greater than or equal to a pre-determined reliability level, namely:

$$P(z_{Wi} \geq z^*_i) \geq \beta_i \quad (i=M, C, B) \qquad (8)$$

where z_{Wi} is the ith attribute in z_W, z^*_i is the minimum acceptable level of the ith attribute, $Pr(z_{Wi} \geq z^*_i)$ is the probability that z_{Wi} exceeds z^*_i and β_i is a reliability level ($0 \leq \beta_i \leq 1$). z^*_i and β_i are selected by the alliance.

Weak sustainability requires the probability that the expected value of a composite index of the attributes exceeds a minimum acceptable value to be greater than or equal to a reliability level, namely:

$$P\{[E(CI_W)] \geq CI^*\} \geq \theta \qquad (9)$$

where $E(CI_W)$ is the expected value of the composite index of the attributes provided by z_W, CI^* is the value of the composite index when each attribute is set equal to its minimum acceptable value and θ is the reliability level established by the alliance where $0 \leq \theta \leq 1$. The weak sustainability criterion is compensatory because high values of one attribute can compensate for low values of another attribute. Probabilities are estimated using simulated or expert-determined values of the attributes. Prato (1998b) provides a more complete discussion of weakly and strongly sustainable resource management.

Suppose the watershed alliance evaluates the strong sustainability of LMP_W for reliability levels of 1, namely $\beta_i=1$ for $i=M$, C, B.

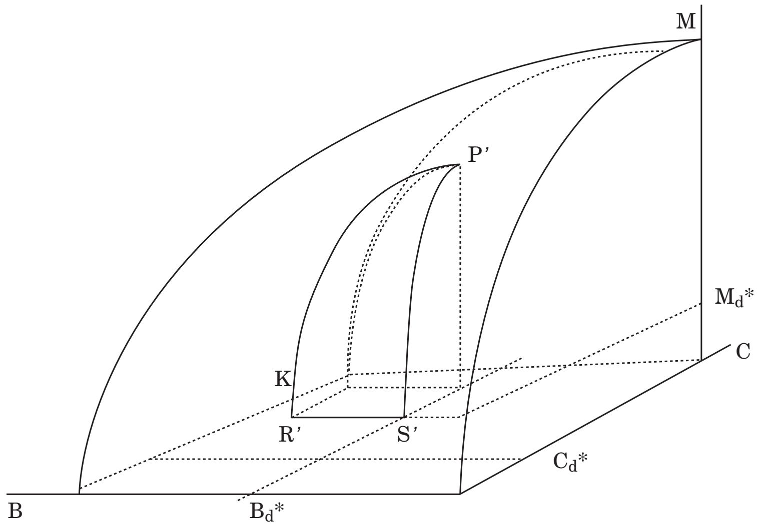

Figure 3. Sustainable efficiency frontier for three attributes: net return (M), soil conservation (C) and biodiversity (B).

This implies that $z_{Wi} \geq z^*_i$ for all attributes. The strongly sustainable combinations of the three attributes constitute the sustainable efficiency frontier. The latter is defined by the surfaces labeled P′R′K and P′R′S′ in Figure 3. Note that P′R′K is a subset of PRK and P′R′S′ is a subset of PRS. Associated with each sustainable combination of attributes is one or more sustainable LMP. If the set of strongly sustainable LMP is designated LMP_F, then LMP_W is strongly sustainable provided $LMP_W \subseteq LMP_F$.

Accounting for uncertainty in evaluating sustainability

While the sustainability assessment described in the previous section accounts for the stochasticity of attributes, it implicitly assumes that the watershed alliance can determine with certainty whether or not a LMP is sustainable. This assumption contradicts the inherent complexity of ecosystems and is likely to result in wrong decisions regarding the sustainability of an LMP. For example, ecologists readily acknowledge they do not know how much biodiversity can be lost before the integrity of ecosystems and human welfare are adversely affected (Barbier *et al*., 1994). Relaxing the certainty assumption would significantly improve sustainability assessments. This is done using Bayes theorem, which provides a statistical rationale for evaluating decision making under risk and uncertainty (Berger, 1985; McDonald and Smith, 1997).

Consider a landscape that can be in one of G states of sustainability, labeled $H_1, \ldots, H_G$ where the degree of sustainability increases from H_1 to H_G. Each state is defined by a set of attribute values. Prior probabilities for the states of sustainability are $P(H_g)$ ($g=1, \ldots, G$). The likelihood that a landscape is unsustainable is high when $P(H_g)$ is high for low values of g. Conversely, the likelihood that the system is sustainable is high when $P(H_g)$ is high for high values of g.

A more precise decision rule is developed. Suppose the watershed alliance enlists a group of experts that divide the $H_1, \ldots, H_G$ states into two sets: H_U and H_S where H_U contains G' states ($G' < G$) deemed to be unsustainable and H_S contains $G - G'$ states deemed to be sustainable. The initial state of the landscape is judged to be unsustainable when $P(H_{g'}) \geq P_u (g'=1, \ldots, G')$ where P_u is a threshold probability determined by the alliance. In other words, the initial state of the landscape is considered to be unsustainable when any of the prior probabilities for unsustainable states is greater than or equal to P_u. Conversely, initial conditions in the landscape are deemed to be sustainable when $P(H_{g'}) < P_u (g'=1, \ldots, G')$. The Bayesian-based sustainability assessment is used below to evaluate polices for achieving sustainable resource management.

Policy analysis

Suppose the initial conditions in a landscape are unsustainable, and the watershed alliance wants to determine the effectiveness of a policy for achieving sustainable resource management. The effectiveness of a policy in achieving sustainability is evaluated as follows. Let z_{d1} be the simulated or expert-determined attributes of the landscape with policy 1. The likelihood function for z_{d1} is $P(z_{d1}|H_l)$. Applying Bayes theorem, the posterior probability that the state is H_g when the attribute values are z_{d1} is:

$$P(H_g|z_{d1}) = [P(H_g)P(z_{d1}|H_g)]/[\Sigma_g P(H_g)P(z_{d1}|H_g)] \quad (10)$$

Posterior probabilities are used to re-evaluate whether or not the conditions in the landscape are sustainable with policy 1. If $P(H_{g'}|z_{d1}) < P_u$, then conditions are sustainable with policy 1. Otherwise, policy 1 does not result in sustainable resource management. The posterior probabilities are calculated for several policies to determine which ones are most likely to result in sustainability. Finally, the most cost-effective policy is selected from among the policies that are most likely to attain sustainable resource management.

Assessing sustainability in a Bayesian framework requires the alliance to estimate the conditional probabilities for different states of nature and the likelihood functions. Conditional probabilities can be established based on expert opinion. The likelihood functions can be estimated using simulated or expert-determined values of the attributes. Estimation of the likelihood functions is more difficult when the attributes are not statistically independent.

Judging the merits of a policy based on Bayes theorem is consistent with the philosophy of adaptive resource management. The basic premise of adaptive resource management is that 'if human understanding of nature is imperfect, then human interactions with nature [e.g. policies] should be experimental' (Lee, 1995:229). This premise implies that experimentation is the primary way of learning about natural systems (Holling, 1978; Walters, 1996). When the ecosystem being evaluated is large and complex, experimentation is not feasible. The MADM applications described in this paper can be combined with Bayes theorem to evaluate beforehand whether or not a particular policy is likely to achieve sustainable resource management. Incorporating Bayes theorem in an interactive decision support system facilitates its application.

Concluding remarks

MADM is useful for identifying preferred management plans for sites and management units in a watershed based on property managers preferences for attributes of efficient management plans and assessing whether implementation of the preferred management plans results in states that are weakly or strongly sustainable at the landscape scale. The MADM application discussed here involves solving a two-stage optimization problem. In the first stage, a stochastic optimization problem is solved for the most efficient combinations of attributes for sites and management units. In the second stage, the decision-maker selects the preferred combination of attributes from among the efficient combinations identified in the first stage. The second-stage decision is modeled by maximizing a risk-neutral utility function that is additive in the attributes of management plans. Associated with each preferred combination of attributes for a site or management unit is a unique management plan referred to as the preferred management plan.

The weak sustainability of the preferred management plans for a landscape is judged by evaluating the probability that the expected value of a composite index of attributes exceeds the minimum acceptable value of that index when the preferred management plans are implemented. Strong sustainability is evaluated by comparing the probability that the expected value of each attribute exceeds the corresponding minimum acceptable value of that attribute when the preferred management plans are implemented. Probabilities are compared to reliability levels selected by a watershed alliance.

Application of the MADM model requires simulated or expert-determined values of the attributes of management plans and estimating or specifying attribute weights (w_i), minimum and maximum acceptable values of attributes (z_i^*) and reliability levels (α_i). These information requirements are substantial especially when there are several management plans and attributes. A property manager's ability to solve the two-stage optimization problem for the preferred combination of attributes and management plans and a watershed alliance's capacity to determine whether management plans are sustainable can be substantially improved by incorporating the MADM model in an interactive decision support system.

Assessment of policies for achieving weakly or strongly sustainable resource conditions is complicated by uncertainty regarding how preferred management plans influence the state of the landscape. To accommodate such uncertainty, the traditional MADM method is modified using Bayes theorem. The resulting procedure allows determination of whether implementation of the preferred management plans is likely to result in a sustainable landscape.

The major management implications of the paper are that: (1) MADM is an appropriate framework for modeling and explaining the selection of management plans at the property and watershed scales; (2) a watershed alliance can use MADM to evaluate the weak and strong sustainability of alternative management plans at the watershed scale; and (3) Bayes theorem provides a useful way to account for uncertainty about whether a particular policy achieve sustainable resource conditions.

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